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Soil Screening Guidance for Radionuclides: Technical Background Document

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DISCLAIMER

Notice: The Soil Screening Guidance for Radionuclides is based on policies set out in the Preamble to the Final Rule of the National Oil and Hazardous Substances Pollution Contingency Plan (NCP), which was published on March 8, 1990 (55 *Federal Register* 8666).

This guidance document sets forth recommended approaches based on EPA's best thinking to date with respect to soil screening for radionuclides. Alternative approaches for screening radionuclides in soil may be found to be more appropriate at specific sites (e.g., where site circumstances do not match the underlying assumptions, conditions, and models of the guidance). The decision whether to use an alternative approach and a description of any such approach should be placed in the Administrative Record for the site.

The policies set out in both the Soil Screening Guidance for Radionuclides: User's Guide and the supporting Soil Screening Guidance for Radionuclides: Technical Background Document are intended solely as guidance to the U.S. Environmental Protection Agency (EPA) personnel; they are not final EPA actions and do not constitute rulemaking. These policies are not intended, nor can they be relied upon, to create any rights enforceable by any party in litigation with the United States government. EPA officials may decide to follow the guidance provided in this document, or to act at variance with the guidance, based on an analysis of specific site circumstances. EPA also reserves the right to change the guidance at any time without public notice.

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PREFACE

This document provides the technical background for the development of methodologies described in the *Soil Screening Guidance for Radionuclides: User's Guide* (EPA/540-R-00-007), along with additional information useful for soil screening. Together, these documents define the framework and methodology for developing Soil Screening Levels (SSLs) for radionuclides commonly found at Superfund sites. This document is similar to a previous guidance document issued by EPA entitled *Soil Screening Guidance: Technical Background Document* (EPA/540/R-95/128), which contains information regarding the soil screening of chemicals commonly found at Superfund sites. This guidance document intended to be consistent with the 1996 guidance document except where it was necessarry to be different based on a technical difference posed by radionuclides.

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Part 1: INTRODUCTION

This document provides the technical background for the Soil Screening Guidance for Radionuclides. The Soil Screening Guidance for Radionuclides is a tool that the U.S. Environmental Protection Agency (EPA) developed to help standardize and accelerate the evaluation and cleanup of soils contaminated with radioactive materials at sites on the National Priorities List (NPL) with anticipated future residential land use scenarios.[1] This guidance provides a methodology for environmental science/engineering professionals to calculate risk-based, site-specific, soil screening levels (SSLs), for radioactive contaminants in soil that may be used to identify areas needing further investigation at NPL sites.

SSLs are not national cleanup standards. SSLs alone do not trigger the need for response actions or define "unacceptable" levels of radionuclides in soil. "Screening," for the purposes of this guidance, refers to the process of identifying and defining areas, radionuclides , and conditions at a particular site that do not require further Federal attention. Generally, at sites where radionuclide concentrations fall below SSLs, no further action or study is warranted under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). (Some States have developed screening numbers or methodologies that may be more stringent than SSLs; therefore further study may be warranted under State programs.) Where radionuclide concentrations equal or exceed the SSLs, further study or investigation, but not necessarily cleanup, is warranted.

EPA's *Soil Screening Guidance: User's Guide* (EPA 1996a) and the *Soil Screening Guidance: Technical Background Document* (EPA 1996b) apply the SSL framework to NPL sites with hazardous organic and inorganic soil contaminants. They do not address sites with radioactive contaminants. These documents provide standardized exposure equations for deriving generic and site-specific SSLs for chemicals under a residential land use setting, assuming three soil exposure pathways—soil ingestion, inhalation of volatiles and fugitive dusts, and ingestion of contaminated ground water. Chemical- specific SSLs are based on a target risk of one-in-a-million (10^{-6}) for carcinogens, a hazard quotient of 1 for noncarcinogens, or, for the ground water migration pathway, a nonzero maximum contaminant level goal (MCLG), maximum contaminant level (MCL), or a risk-based level. For each contaminant, the lowest pathway-specific SSL is selected as the appropriate screening level. The guidance provides default, generic SSLs for 110 chemicals based on parameter values consistent with Superfund's concept of reasonable maximum exposure (RME). The guidance also presents a tiered approach for evaluating contaminant migration to ground water, methods for deriving site-specific dilution/attenuation factors (DAFs) and particulate emission factors (PEFs), and guidelines for measuring and comparing contaminant soil concentrations with SSLs.

A soil screening framework for radionuclides has been developed that is consistent and compatible with the SSL framework for chemicals. Since radiologically contaminated sites are a subset of hazardous waste sites, much of the analysis conducted for the two 1996 SSL documents was considered valid for this effort to address radioactive contaminants. Radionuclide SSLs are based on a target risk of one-in-a-million (10^{-6}), or, for the ground water migration pathway, a maximum contaminant level (MCL). These SSLs, in activity units of picocuries per gram of soil (pCi/g), are derived from equations combining exposure information assumptions with EPA radiotoxicity data. The Soil Screening Guidance provides a framework for screening soils contaminated with radionuclides that encompasses both simple and more detailed approaches for calculating site-specific SSLs, and generic SSLs for use where site-specific data are limited. The *Soil Screening Guidance for Radionuclides: User's Guide* (U.S. EPA, 2000)

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1. Note that the Superfund program defines "soil" as having a particle size under 2 millimeters, while the RCRA program allows for particles under 9 millimeters in size.

focuses on the application of the simple site-specific approach by providing a step-by-step methodology to calculate site-specific SSLs and plan the sampling necessary to apply them. This Technical Background Document describes the development and technical basis of the methodology presented in the User's Guide. It includes detailed modeling approaches for developing screening levels that can take into account more complex site conditions than the simple site-specific methodology emphasized in the User's Guide. It also provides generic SSLs for the most common contaminants found at NPL sites.

1.1 Background

The Soil Screening Guidance for chemicals was the result of technical analyses and coordination with numerous stakeholders. The effort began in 1991 when the EPA Administrator charged the Office of Solid Waste and Emergency Response (OSWER) with conducting a 30-day study to outline options for accelerating the rate of cleanups at NPL sites. One of the specific proposals of the study was for OSWER to "examine the means to develop standards or guidelines for contaminated soils." From 1991 to 1995, several drafts of the guidance and the accompanying technical background document received widespread reviews both within and outside EPA. In the Spring of 1995, final drafts were released for public comment and external scientific peer review. Many reviewers' comments contributed significantly to the development of this flexible tool that uses site-specific data in a methodology that can be applied consistently across the nation.

These SSL guidance documents specifically excluded radionuclides. ORIA developed a soil-screening guidance document for radionuclides to supplement 1996 guidance. The SSG for radionuclides guidance and TBD are intended to be consistent with the original chemical SSG documents, except where there is a technical difference between chemicals and radionuclides. Almost all of the new information contained in the radionuclide SSG guidance and TBD have been previously released for public comment and/or externally scientific peer reviewed as part of other guidance/rulemaking efforts.

Since radionuclides are considered a hazardous substance under section 101(14)(E) of CERCLA, the Agency has the authority and responsibility to oversee the cleanup of Superfund sites contaminated with radionuclides. This guidance will assist OSC's and RPM's in making decisions at these sites.

1.2 Purpose of SSLs

In identifying and managing risks at sites, EPA considers a spectrum of radionuclide concentrations. The level of concern associated with those concentrations depends on the likelihood of exposure to radioactive soil contamination at levels of potential concern to human health. Figure 1 illustrates the spectrum of soil contamination encountered at Superfund sites and the conceptual range of risk management. At one end are levels of contamination that clearly warrant a response action; at the other end are levels that warrant no further study under CERCLA. Appropriate cleanup goals for a particular site may fall anywhere within this range depending on site-specific conditions. Screening levels identify the lower bound of the spectrum—levels below which no further study is warranted under CERCLA, provided conditions associated with the SSLs are met.

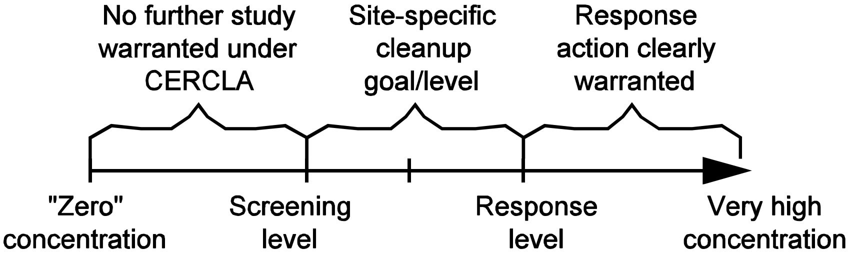

Figure 1. Conceptual Risk Management Spectrum for Contaminated Soil

Although the application of SSLs during site investigations is not mandatory at sites being addressed by CERCLA or RCRA, EPA recommends the use of SSLs as a tool to facilitate prompt identification of radionuclides and exposure areas of concern. EPA developed the Soil Screening Guidance for Radionuclides to be consistent with and to enhance the current Superfund investigation process and anticipates it will be used primarily during the early stages of a remedial investigation (RI) at NPL sites. It does not replace the Remedial Investigation/Feasibility Study (RI/FS) or risk assessment, but use of screening levels can focus the RI and risk assessment on aspects of the site that are more likely to be a concern under CERCLA. By screening out areas of sites, potential radionuclides of concern, or exposure pathways from further investigation, site managers and technical experts can limit the scope of the remedial investigation or risk assessment. SSLs can save resources by helping to determine which areas do not require additional Federal attention early in the process. Furthermore, data gathered during the soil screening process can be used in later Superfund phases, such as the baseline risk assessment, feasibility study, treatability study, and remedial design. This guidance may also be appropriate for use by the removal program when demarcation of soils above residential risk-based numbers coincides with the purpose and scope of the removal action. EPA created the Soil Screening Guidance for Radionuclides to be consistent with and to enhance current Superfund processes.

The purpose of the process presented in this guidance is to develop and apply simple, site-specific soil screening levels. This approach is likely to be most useful where it is difficult to determine whether areas of soil are contaminated to an extent that warrants further investigation or response (e.g., whether areas of soil at an NPL site require further investigation under CERCLA through an RI/FS). The screening levels have been developed assuming future residential land use assumptions and related exposure scenarios. Although some of the models and methods presented in this guidance could be modified to address exposures under other land uses, EPA has not yet standardized assumptions for those other uses. Using this guidance for sites where residential land use assumptions do not apply could result in overly conservative screening levels. However, EPA recognizes that some parties responsible for sites with non-residential land use might still benefit from using SSLs as a tool to conduct conservative initial screening.

EPA created the *Soil Screening Guidance for Radionuclides: User's Guide* (U.S. EPA, 1999) to be easy to use: it provides a simple step-by-step methodology for calculating SSLs that are specific to the user's site. Applying site-specific screening levels involves developing a conceptual site model (CSM), collecting a few easily obtained site-specific soil parameters (such as the dry bulk density and percent soil moisture), and sampling soil to measure radionuclide levels in surface and subsurface soils. Often, much of the information needed to develop the CSM can be derived from previous site investigations (e.g., the preliminary assessment/site inspection [PA/SI]) and, if properly planned, SSL sampling can be accomplished in one mobilization.

SSLs can be used as Preliminary Remediation Goals (PRGs) provided appropriate conditions are met (i.e., conditions found at a specific site are similar to conditions assumed in developing the SSLs). The concept of calculating risk-based soil levels for use as PRGs (or "draft" cleanup levels) was introduced in the *Risk Assessment Guidance for Superfund* (RAGS), *Volume I, Human Health Evaluation Manual* (HHEM), *Part B* (U.S. EPA, 1991b). PRGs are risk-based values that provide a reference point for establishing site-specific cleanup levels. The models, equations, and assumptions presented in the Soil Screening Guidance for Radionuclides and described herein supersede those described in RAGS HHEM, Part B, for residential soils. **In addition, this guidance presents methodologies to address the leaching of radionuclides through soil to an underlying potable aquifer. This pathway should also be addressed in the development of PRGs.**

EPA emphasizes that SSLs are **not** cleanup standards. SSLs should not be used as site-specific cleanup levels unless a site-specific nine-criteria evaluation using SSLs as PRGs for soils indicates that a selected remedy achieving the SSLs is protective, compliant with applicable or relevant and appropriate requirements (ARARs), and appropriately balances the other criteria, including cost. PRGs may then be converted into final cleanup levels based on the nine-criteria analysis described in the National Contingency Plan (NCP; Section 300.430 (3)(2)(A)). The directive entitled *Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions* (U.S. EPA, 1991c) discusses the modification of PRGs to generate cleanup levels.

The generic SSLs provided in Appendix A are calculated from the same equations used in the simple site-specific methodology, but are based on a number of default assumptions chosen to be protective of human health for most site conditions. Generic SSLs can be used in place of site-specific screening levels; however, they are expected to be generally more conservative than site-specific levels. The site manager should weigh the cost of collecting the data necessary to develop site-specific SSLs with the potential for deriving a higher SSL that provides an appropriate level of protection.

1.3 Scope of Soil Screening Guidance for Radionuclides

The Soil Screening Guidance for Radionuclides incorporates readily obtainable site data into simple, standardized equations to derive site-specific screening levels for selected radionuclides and exposure pathways. Key attributes of the Soil Screening Guidance for Radionuclides are given in Highlight 1.

Highlight 1: Key Attributes of the Soil Screening Guidance for Radionuclides

- Standardized equations are presented to address human exposure pathways in a residential setting consistent with Superfund's concept of "Reasonable Maximum Exposure" (RME).
- Source size (area and depth) can be considered on a site-specific basis.
- Parameters are identified for which site-specific information is needed to develop site-specific SSLs.
- Default parameter values are provided to calculate generic SSLs where site-specific information is not available.
- SSLs are generally based on a 10^{-6} lifetime cancer risk. SSLs for migration to ground water are based on maximum contaminant levels (MCLs).
- Radiation risk coefficients used to calculate SSLs represent the average risk per unit exposure to members of a population exposed throughout life to a constant concentration of a radionuclide in a specific environmental medium. They assume no radioactive decay.

1.3.1 Exposure Pathways.

In a residential setting, potential pathways of exposure to radionuclides in soil included in this guidance are as follows (see Figure 2):

- Direct ingestion of soil
- Inhalation of fugitive dusts
- External radiation exposure from photon-emitting radionuclides in soil

- Ingestion of homegrown produce that has been contaminated via plant uptake

- Ingestion of contaminated ground water caused by migration of radionuclides through soil to an underlying potable aquifer

The Soil Screening Guidance for Radionuclides addresses each of these pathways to the greatest extent practical. The mode of exposure to radionuclides is different than that of chemicals. This renders some chemical pathways inapplicable to radionuclides (e.g., inhalation of volatiles, dermal absorption) while adding other pathways unique to radiation (e.g., external exposure to photons emitted by radionuclides). The radiological pathways listed above represent the most likely exposure mechanisms for individuals in a residential setting. The external exposure pathway is, for most radionuclides, the dominant exposure and typically represents the most significant risk. For some radionuclides, the ingestion of contaminated produce and drinking water constitute the most likely exposure pathways provided that these items are obtained from onsite sources. The inhalation of fugitive dust pathway is included in the analysis; however, it is of significance for only a very few radionuclides. All of these pathways have generally accepted radiological risk methods, models, and assumptions that lend themselves to a standardized approach.

The Soil Screening Guidance for Radionuclides addresses the human exposure pathways listed previously and will be appropriate for most residential settings. The presence of additional pathways or unusual site conditions does not preclude the use of SSLs in areas of the site that are currently residential or likely to be residential in the future. However, the risks associated with these additional pathways or conditions (e.g., fish consumption, raising of livestock for meat or milk consumption, fugitive dusts caused by heavy truck traffic on unpaved roads) should be considered in the remedial investigation/feasibility study (RI/FS) to determine whether SSLs are adequately protective.

An ecological assessment should also be performed as part of the RI/FS to evaluate potential risks to ecological receptors.

The Soil Screening Guidance for Radionuclides should not be used for areas with chemical contaminants.

1.3.2 Exposure Assumptions. SSLs are risk-based concentrations derived from equations combining exposure assumptions with EPA radiotoxicity data. The models and assumptions used to calculate SSLs were developed to be consistent with Superfund's concept of "reasonable maximum exposure" (RME) in the residential setting. The Superfund program's method to estimate the RME for chronic exposures on a site-specific basis is to combine an average exposure point concentration with reasonably conservative values for intake and duration in the exposure calculations (U.S. EPA, 1989b; U.S. EPA, 1991a). The default intake and duration assumptions presented in U.S. EPA (1991a) were chosen to represent individuals living in a small town or other nontransient community. (Exposure to members of a more transient community is assumed to be shorter and thus associated with lower risk.) Exposure point concentrations are either measured at the site (e.g., ground water concentrations at a receptor well) or estimated using exposure models with site-specific model inputs. An average concentration term is used in most assessments where the focus is on estimating long-term, chronic exposures. Where the potential for acute toxicity is of concern, exposure estimates based on maximum concentrations may be more appropriate.

The resulting site-specific estimate of RME is then compared with a radionuclide-specific toxicity criterion. EPA recommends using criteria from the Health Effects Assessment Summary Tables (HEAST) which may be found on the internet at the following address: http://www.epa.gov/superfund/programs/risk/calctool.htm, although values from other sources may be used in appropriate cases.

SSLs are concentrations of radionuclides in soil that are designed to be protective of exposures in a residential setting. A site-specific risk assessment is an evaluation of the risk posed by exposure to site radionuclides in various media. To calculate SSLs, the exposure equations and pathway models are run in reverse to backcalculate an "acceptable level" of a radionuclide in soil corresponding to a specific level of risk.

1.3.3 Risk Level.

For each pathway, radiotoxicity criteria are used to define an acceptable level of radionuclides in soil, based on a one-in-a-million (10^{-6}) individual excess lifetime cancer risk. SSLs are backcalculated for migration to ground water pathway using ground water concentration limits [maximum contaminant levels (MCLs)].

The potential for additive effects has not been "built in" to the SSLs through apportionment. While the pathways included in the analysis are considered to represent those a residential setting, SSLs are not calculated for a specific scenario (i.e., SSLs are not summed over a set of pathways). EPA believes that setting a 10^{-6} risk level for individual radionuclides and pathways will generally lead to cumulative risks within the risk range (10^{-4} to 10^{-6}) for the combinations of radionuclides typically found at Superfund sites.

1.3.4 SSL Model Assumptions.

The models used to calculate SSLs were designed for use at an early stage of site investigation when site information may be limited. Because of this constraint, they incorporate a number of simplifying assumptions.

The models assume that the source is infinite (steady over time). Although the assumption is highly conservative, a finite source model cannot be applied unless there are accurate data regarding source size and volume. EPA believes it to be unlikely that such data will be available from the limited subsurface sampling that is done to apply SSLs. However, EPA also recognizes that infinite source (i.e., steady-state) models can violate mass balance (i.e., can release more contaminants than are present) for certain contaminants and site conditions (e.g., small sources). To address this problem, this guidance includes simple models that provide a mass-based limit for the inhalation and migration to ground water SSLs (see Section 2.6). **A site-specific estimate of source depth and area are required to calculate SSLs using these alternative models**.

The infinite source (i.e., steady-state) assumption leads to several other simplifying assumptions. Fractionation of contaminant mass between the inhalation and migration to ground water pathways cannot be addressed with infinite source (i.e., steady-state) models. For the migration to ground water pathway, an infinite source (i.e., steady-state) overrides adsorption in the unsaturated zone or in the aquifer. The models also assume that contamination is evenly distributed throughout the source (i.e., homogeneous) and that no biological or chemical degradation occurs in the soil or in the aquifer. Again, models capable of addressing heterogeneities or degradation processes require collection of site-specific data that is well beyond the scope of the Soil Screening Guidance for Radionuclides.

Although the Soil Screening Guidance for Radionuclides encourages the use of site-specific data to calculate SSLs, conservative default parameters are provided for use where site-specific data are not available. These defaults are described in Part 2 of this document. Appendix A provides an example set of "generic" SSLs for 60 radionuclides that are calculated using these defaults. Because they are designed to be protective of most site conditions across the nation, they are conservative.

A default 0.5 acre source area is used to calculate the generic SSLs. A 30 acre source size was used in the December 1994 Soil Screening Guidance for chemicals. EPA received an overwhelming number of comments that suggest that most contaminated soil sources addressed under the Superfund program are 0.5 acres or smaller. Because of the infinite source (i.e., steady-state) assumption, generic SSLs based on a 0.5 acre source size can be protective of larger sources as well (see Appendix A). However, this hypothesis should be examined on a case-by-case basis before applying the generic SSLs to sources larger than 0.5 acre.

1.4 Organization of the Document

Part 2 of this document describes the development of the simple equations used to calculate SSLs. It describes and supports the assumptions behind these equations and presents the results of analyses conducted to develop the SSL methodology. Some of the more sensitive parameters are identified for which site-specific data are likely to have a significant impact. Default values are provided along with their sources and limitations.

Part 3 presents information on other, more complex models that can be used to calculate SSLs when more extensive site data are available or can be obtained. Some of these models can consider a finite source and fractionation between exposure pathways. They also can model more complex site conditions than the simple SSL equations, including conditions that can lead to higher, yet still protective, SSLs (e.g., thick unsaturated zones, biological and chemical degradation, layered soils).

Part 4 provides the technical background for the development of the soil sampling design methodology for SSL application. It addresses methods for surface soil, including a test based on a maximum soil composite sample, the Max test, and the Sign test, which allows decision errors to be controlled. Lastly, Part 4 provides simulation results that measure the performance of these methods and sample size tables for different contaminant distributions and compositing schemes. Step-by-step guidance is provided for developing sample designs using each statistical procedure.

Part 5 describes the selection and development of the radionuclide properties used to calculate SSLs.

Part 2: DEVELOPMENT OF PATHWAY-SPECIFIC SOIL SCREENING LEVELS

This part of the Technical Background Document for Radionuclides describes the methods used to calculate SSLs for residential exposure pathways, along with their technical basis and limitations associated with their use. Simple, standardized equations have been developed for the more common radiological exposure pathways at Superfund sites:

- Direct ingestion of soil (Section 2.2)
- Inhalation of fugitive dust (Section 2.3)
- External radiation exposure from photon-emitting radionuclides in soil (Section 2.4)
- Ingestion of homegrown produce that has been contaminated via plant uptake (Section 2.5)
- Ingestion of contaminated ground water caused by migration of radionuclides through soil to an underlying potable aquifer (Section 2.6).

The equations were developed under the following constraints:

- They should be consistent with current Superfund risk assessment methodologies and guidance.
- To be appropriate for early-stage application, they should be simple and easy to apply.
- They should allow the use of site-specific data where they are readily available or can be easily obtained.
- The process of developing and applying SSLs should generate information that can be used and built upon as a site evaluation progresses.

The equations include easily obtained site-specific input parameters. Conservative default values have been developed for use where site-specific data are not available. Generic SSLs, calculated for 60 radionuclides using these default values, are presented in Appendix A. The generic SSLs are conservative, since the default values are designed to be protective at most sites across the country.

The migration to ground water pathway equation assumes an infinite source (e.g., source remains constant over time). As pointed out by several commenters to the December 1994 draft Soil Screening Guidance for chemicals (U.S. EPA, 1994h), SSLs developed using these models may violate mass-balance for certain contaminants and site conditions (e.g., small sources). To address this concern, EPA has incorporated a simple mass-limit model for this pathway assuming that the entire volume of contamination leaches over the duration of exposure and that the level of contaminant at the receptor does not exceed the health-based limit (Section 2.7). **Because they require a site-specific estimate of source depth, these alternative models cannot be used to calculate generic SSLs.**

The mode of exposure to radionuclides is different than that of chemicals. This renders some chemical pathways inapplicable to radionuclides (e.g., inhalation of volatiles, dermal absorption) while adding other pathways unique to radiation (e.g., external exposure to photons emitted by radionuclides). The radiological pathways listed above represent the most likely exposure mechanisms for individuals in a residential setting. For soils under the residential land use assumption, the external exposure pathway will typically be the dominant exposure pathway for most radionuclides (e.g., ^{60}Co, ^{137}Cs, ^{226}Ra, ^{238}U). For a good number of radionuclides (e.g., ^{14}C, ^{63}Ni, ^{90}Sr, ^{99}Tc), the plant ingestion pathway often dominates, although not to the extent that the external exposure pathway does. The soil ingestion pathway also plays a dominant role for some radionuclides of interest (i.e., ^{129}I, ^{232}Th, ^{239}Pu, ^{241}Am). In the majority of cases, the inhalation of fugitive dust pathway and the migration to groundwater pathway play an insignificant role.

In addition to the more common pathways of exposure in a residential setting, concerns have been raised regarding the potential for migration of radon from subsurface soils into basements. The dominant factor in indoor radon levels is home construction practices and the extent to which these practices employ radon-resistant techniques. Homes built atop soil with identical levels of radium can have orders of magnitude differences in indoor radon levels depending on the extent to which radon-resistant techniques are used. As a naturally-occurring radionuclide, radium is present in all soils. Reducing the radium content in the soil may not result in any reduction in indoor radon levels. However, taking simple and inexpensive steps in home construction will ensure that radon levels in homes are kept below Applicable or Relevant and Appropriate Requirements (ARARs) levels. For existing homes with elevated levels of radon, a variety of methods can be used to reduce radon concentrations to ARAR levels. Discussion of radon mitigation standards may be found in several EPA publications, including *Radon Mitigation Standards*, EPA 402-R-93-078 (U.S. EPA, 1994i). Also note that potential ARARs exist for indoor radon under 192.12(b)(1) and 192.41(b). For further guidance on using these ARARs, see the August 1997 memorandum from Stephen Luftig (OERR) and Larry Weinstock (ORIA) titled "Establishment of cleanup levels for CERCLA sites with radioactive contamination," OSWER Directive 9200.4-18, (U.S. EPA, 1997a).

Section 2.1 describes the human health basis of the Soil Screening Guidance for Radionuclides and provides the human toxicity and health benchmarks necessary to calculate SSLs. The selection and development of the radiological properties required to calculate SSLs are described in Part 5 of this document.

2.1 Human Health Basis

The 60 radionuclides for which generic SSLs have been calculated are listed in Table 2.1. Principal radionuclides are radionuclides with half-lives greater than six months. The decay products of any principal radionuclide down to, but not including, the next principal radionuclide in its decay chain are called associated radionuclides and consist of radionuclides with half-lives less than six months. It is assumed that a principal radionuclide is in secular equilibrium with its associated radionuclides at the point of exposure. This assumption is reasonable because it usually takes about three years or longer to clean up a site. Associated decay chains are indicated, as well as principal radionuclide half-life and the terminal nuclide or radionuclide (i.e., the principal radionuclide or stable nuclide that terminates an associated decay chain).

Tables 2.2 and 2.3 list the regulatory and human health benchmarks necessary to calculate SSLs for 60 radionuclides. For all pathways other than migration to ground water, these benchmarks are based on the latest available radionuclide cancer slope factors. For migration to ground water, drinking water MCLs are shown.

The human health benchmarks in Table 2.2 were obtained from *Health Risks from Low-Level Environmental Exposure to Radionuclides, Federal Guidance Report No. 13, Part I - Interim Version*, EPA 402-R-97-014 (U.S. EPA 1998a) (also know as FGR 13). When the slope factors are not available in FGR 13, data are taken from *Health Effects Assessment Summary Tables (HEAST): Annual Update, FY-1995*, (U.S. EPA, 1995a) (also known as the

HEAST report). Slope factors are listed for each route of intake for principal radionuclides in units of picocuries (pCi).[1] Radionuclides are presented alphabetically by element and atomic weight.

MCLs in Table 2.3 were obtained from *Drinking Water Regulations and Health Advisories* (U.S. EPA, 1995a). The current MCLs for radionuclides were enacted in 1976 and are set at 4 mrem/yr for the sum of the doses from beta particles and photon emitters, 15 pCi/L for gross alpha particle activity (including Ra-226, but excluding uranium and radon), and 5 pCi/L combined for Ra-226 and Ra-228. In July 1991, EPA proposed to revise the MCLs for Ra-226 and Ra-228 to 20 pCi/L for each, change the methodology used for determining a 4 mrem/yr dose for the sum of the doses from beta particles and photon emitters, alter the definition of alpha particle activity to exclude Ra-226, and establishing new MCLs of 300 pCi/L for Rn-222 and 20 µg/L (30 pCi/L) for uranium (56 FR 33050). EPA is under Court Order to either finalize the 1991 proposal for radionuclides (except for radon), or to ratify existing standards by November 2000. On April 21, 2000 EPA solicited comment in a Notice of Data Availability (NODA) on three options for a uranium MCL: 1) 20 ug/l **and** 20 pCi/l as a preferred option, 2) 40 ug/l **and** 40 pCi/l, and 3) 80 ug/l **and** 80 pCi/l (65 FR 21576). In this NODA, EPA indicated that changes would not be made to the existing MCLs for radium, alpha particle activity, and beta particles and photon emitters. The 1996 Amendments to the Safe Drinking Water Act (SDWA) require EPA to propose a MCL for radon by August 1999, and to finalize the MCL by August 2000. To comply with the requirements of the amended SDWA, on August 6, 1997, EPA withdrew its 1991 proposal for Rn-222 (62 FR 42221). EPA issued a new proposal for Rn-222 on November 2, 1999 (64 FR 59246). EPA proposed an MCL of 300 pCi/l with an alternative MCL of 4,000 pCi/l if a state or local indoor radon mitigation program was established.

References for each table are updated regularly. **Prior to calculating SSLs, the values in Tables 2.2 and 2.3 should be checked against the most recent version of these sources to ensure that they are up-to-date.**

Selected radionuclides and radioactive decay chain products are designated with the suffix "+D" (e.g., U-238+D, Ra-226+D, Cs-137+D) to indicate that cancer risk estimates for these radionuclides include the contributions from their short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal or parent nuclide in the environment.[2] In most cases, site-specific analytical data should be used to establish the actual degree of equilibrium between each parent radionuclide and its decay products in each media sampled. However, in the absence of empirical data, the "+D" values for radionuclides should be used unless there are compelling reasons not to. Note that there may be circumstances, such as long disposal times or technologically enhanced concentrations of naturally occurring radionuclides, that may necessitate the combination of the risks of a parent radionuclide and its decay products over several contiguous subchains. For example, Ra-226 soil analyses at a site might show that all radium decay products are present in secular equilibrium down to stable Pb-206. In this case, Ra-226 risk calculations should be based on the ingestion, inhalation and external exposure slope factors for the Ra-226+D subchain, plus the ingestion, inhalation and external exposure factors for the Pb-210+D subchain. For actual sites, users should consult with a health physicist or radiochemist (1) to evaluate the site-specific analytical data to determine the degree of equilibrium between parent radionuclides and decay members of contiguous decay chains and (2) to assist in the combination of appropriate slope factor values.

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[1] Slope factors are reported in the customary units of risk per picocurie (1 pCi = 10^{-12} curies (Ci) = 3.7×10^{-2} nuclear transformations per second) for consistency with the system used for radionuclides in the IRIS database. If required, slope factors can be converted into the International System (SI) units of becquerels (1 Bq = 1 nuclear transformation per second) by multiplying each inhalation, ingestion, or external exposure value by 27.03. Users can calculate cancer risks using slope factors expressed in either customary units or SI units with equivalent results, provided that they also use air, water and soil concentration values in the same system of units.

[2] There is one exception to the assumption of secular equilibrium. For the inhalation slope factor for Rn-222+D reported in the table, ORIA assumes a 50% equilibrium value for radon decay products (Po-218, Pb-214, Bi-214 and Po-214) in air.

Table 2.1. Radionuclides Included in Generic Soil Screening Analysis

Principal Radionuclide[a]		Associated Decay Chain[b]	Terminal Nuclide or Radionuclide[c]	
Nuclide	**Half-life (yr)**		**Nuclide**	**Half-life (yr)**
Ac-227+D	22	[Th-227 (98.6%, 19 d) Fr-223 (1.4%, 22 min)] Ra-223 (11 d) Rn-219 (4 s) Po-215 (2 ms) Pb-211 (36 min) Bi-211 (2 min) [Tl-207 (99.7%, 5 min) Po-211 (0.3%, 0.5 s)]	Pb-207	stable
Ag-108m	127	-	Pd-108 (91%) [Cd-108 (98%} Ag-108 (9%) Pd-108 (2%)]	stable stable 2 min stable
Ag-110m	0.7	-	Cs-110 (99%) [Cd-110 (99.7%) Ag-110 (1%) Pd-110 (0.3%)]	stable stable 25 s stable
Am-241	432	-	Np-237	2.1E+6
Am-243+D	7,400	Np-239 (2 d)	Pu-239	2.4E+4
Bi-207	38	-	Pb-207	stable
C-14	5,730	-	N-14	stable
Cd-109	1.3	-	Ag-109	stable
Ce-144+D	0.8	[Pr-244 (9%, 17 min) Pr-244m (2%, 7 min)]	Nd-144	stable
Cl-36	3.0E+5	-	S-36	stable
Cm-243	28	-	Am-243 (0.2%)[e]	7,400
Cm-244	18	-	Pu-240	6,600
Co-57	0.7	-	Fe-57	stable
Co-60	5	-	Ni-60	stable
Cs-134	2	-	Ba-134 (~100%)	stable
Cs-135	3E+6	-	Ba-135	stable
Cs-137+D	30	Ba-137m (95%, 3 min)	Ba-137	stable
Eu-152	13	-	Sm-152 (72%) Gd-152 (28%)	stable 1.1E+14
Eu-154	8	-	Gd-154 (~100%)	stable
Eu-155	5	-	Gd-155	stable
Fe-55	3	-	Mn-55	stable
Gd-153	0.7	-	Eu-153	stable
H-3	12	-	He-3	stable
I-129	1.6E+7	-	Xe-129	stable
K-40	1.3E+9	-	Ca-40 (89%) Ar-40 (11%)	stable
Mn-54	0.9	-	Cr-54	stable

Table 2.1. Radionuclides Included in Generic Soil Screening Analysis

Principal Radionuclide[a]		Associated Decay Chain[b]	Terminal Nuclide or Radionuclide[c]	
Nuclide	Half-life (yr)		Nuclide	Half-life (yr)
Na-22	3	-	Ne-22	stable
Nb-94	2.0E+4	-	Mo-94	stable
Ni-59	7.5E+4	-	Co-59	stable
Ni-63	100	-	Cu-53	stable
Np-237+D	2.1E+6	Pa-233 (27 d)	U-233	1.6E+5
Pa-231	3.3E+4	-	Ac-227	22
Pb-210+D	22	Bi-210 (5 d) Po-210 (138 y)	Pb-206	stable
Pm-147	3	-	Sm-147	1.1E+11
Pu-238	88	-	U-234	2.4E+5
Pu-239	2.4E+4	-	U-235	7E+8
Pu-240	6,500	-	U-236	2.3E+6
Pu-241	14	-	Am-241	432 y
Pu-242	3.8E+5	-	U-238	4.5E+9
Pu-244+D	9.3E+7	U-240 ~100%, 14) Np-240	Pu-240	6,500
Ra-226+D	1,600	Rn-222 (4 d) Po-218 (3 min) Pb-214 (~100%, 27 min) Bi-214 (20 min) Po-214 (~100%, 1 min)	Pb-210	22
Ra-228+D	8	Ac-228 (6 h)	Th-228	2
Ru-106+D	1	Rh-106 (30 s)	Pd-106	stable
Sb-125+D	3	Te-125m (23%, 58 d)	Te-125	stable
Sm-147	1.1E+11	-	Nd-143	stable
Sm-151	90	-	Eu-151	stable
Sr-90+D	29	Y-90 (64 h)	Zr-90	stable
Tc-99	2.1E+5	-	Ru-99	stable
Th-228+D	2	Ra-224 (4 d) Rn-220 (56 s) Po-216 (0.2 s) Pb-212 (11h) Bi-212 (61 min) [Po-212 (64%, 0.3 µs) Tl-208 (36%, 3 min)]	Pb-208	stable
Th-229+D	7,300	Ra-225 (15 d) Ac-225 (10 d) Fr-221 (5 min) At-217 (32 ms) Bi-213 (46 min) [Po-213 (98%, 4 µs) Tl-209 (2%, 2 min)] Pd-209 (3 h)	Bi-209	stable
Th-230	7.7E+4	-	Ra-226	1,600
Th-232	1.4E+10	-	Ra-228	6

Table 2.1. Radionuclides Included in Generic Soil Screening Analysis

Principal Radionuclide[a]		Associated Decay Chain[b]	Terminal Nuclide or Radionuclide[c]	
Nuclide	Half-life (yr)		Nuclide	Half-life (yr)
Tl-204	4	-	Pb-204 (97%) Hg-204 (3%)	stable stable
U-232	72	-	Th-228	2
U-233	1.6E+5	-	Th-229	7,300
U-234	2.4E+5	-	Th-230	8E+4
U-235+D	7.0E+8	Th-231 (26 h)	Pa-231	3.4E+4
U-236	2.3E+6	-	Th-232	1.4E+10
U-238+D	4.5E+9	Th-234 (24 d) [Pa-234m (99.8%, 1 min) Pa-234 (0.2%, 7 h)]	U-234	2.4E+5
Zn-65	0.7	-	Cu-65	stable

[a] Radionuclides with half-lives greater than six months. "+D" designates principal radionuclides with associated decay chains.

[b] The chain of decay products of a principal radionuclide extending to (but not including) the next principal radionuclide or a stable nuclide. Half-lives are given in parentheses. Branches are indicated by square brackets with branching ratios in parentheses.

[c] The principal radionuclide or stable nuclide that terminates an associated decay chain.

[d] A hyphen indicates that there are no associated decay products.

[e] The branching decay for Pu-241 and Cm-243 involves multiple principal radionuclides and associated radionuclides.

Table 2.2. Radionuclide Cancer Morbidity - Slope Factors (1)

	Slope Factor (Morbidity Risk Coefficient) Lifetime Excess Cancer Risk per Unit Exposure					
Radionuclide	Water Ingestion (risk/pCi)	Food Ingestion (risk/pCi)	Soil Ingestion (risk/pCi)	Inhalation (risk/pCi)	External Exposure (risk/yr per PCi/g soil)	Notes
Ac-227+D	4.86E-10	6.53E-10	1.16E-09	2.09E-07	1.47E-06	2
Ag-108m	8.14E-12	1.12E-11	1.92E-11	2.67E-11	7.19E-06	2
Ag-110m	9.88E-12	1.37E-11	2.37E-11	2.83E-11	1.30E-05	2
Am-241	1.04E-10	1.34E-10	2.17E-10	2.81E-08	2.76E-08	
Am-243+D	1.08E-10	1.42E-10	2.32E-10	2.70E-08	6.36E-07	2
Bi-207	5.66E-12	8.14E-12	1.49E-11	2.10E-11	7.08E-06	
C-14	1.55E-12	2.00E-12	2.79E-12	7.07E-12	7.83E-12	3
Cd-109	5.00E-12	6.70E-12	1.14E-11	2.19E-11	8.73E-09	
Ce-144+D	3.53E-11	5.19E-11	1.02E-10	1.10E-10	2.44E-07	2
Cl-36	3.30E-12	4.44E-12	7.66E-12	2.50E-11	1.74E-09	
Cm-243	9.47E-11	1.23E-10	2.05E-10	2.69E-08	4.19E-07	
Cm-244	8.36E-11	1.08E-10	1.81E-10	2.53E-08	4.85E-11	
Co-57	1.04E-12	1.49E-12	2.78E-12	2.09E-12	3.55E-07	
Co-60	1.57E-11	2.23E-11	4.03E-11	3.58E-11	1.24E-05	
Cs-134	4.22E-11	5.14E-11	5.81E-11	1.65E-11	7.10E-06	
Cs-135	4.74E-12	5.88E-12	7.18E-12	1.86E-12	2.36E-11	
Cs-137+D	3.04E-11	3.74E-11	4.33E-11	1.19E-11	2.55E-06	2
Eu-152	6.07E-12	8.70E-12	1.62E-11	9.10E-11	5.30E-06	
Eu-154	1.03E-11	1.49E-11	2.85E-11	1.15E-10	5.83E-06	
Eu-155	1.90E-12	2.77E-12	5.40E-12	1.48E-11	1.24E-07	
Fe-55	8.62E-13	1.16E-12	2.09E-12	7.99E-13	0	
Gd-153	1.52E-12	2.22E-12	4.26E-12	6.55E-12	1.62E-07	
H-3	5.07E-14	6.51E-14	9.25E-14	5.62E-14	0	4
I-129	1.48E-10	3.22E-10	2.71E-10	6.07E-11	6.10E-09	5
K-40	2.47E-11	3.43E-11	6.18E-11	1.03E-11	7.97E-07	
Mn-54	2.28E-12	3.11E-12	5.14E-12	5.88E-12	3.89E-06	
Na-22	9.62E-12	1.26E-11	1.97E-11	3.89E-12	1.03E-05	
Nb-94	7.77E-12	1.11E-11	2.05E-11	3.77E-11	7.29E-06	
Ni-59	2.74E-13	3.89E-13	7.33E-13	4.66E-13	0	
Ni-63	6.70E-13	9.51E-13	1.79E-12	1.64E-12	0	
Np-237+D	6.74E-11	9.10E-11	1.62E-10	1.77E-08	7.97E-07	2
Pa-231	1.73E-10	2.26E-10	3.74E-10	4.55E-08	1.39E-07	
Pb-210+D	1.27E-09	3.44E-09	2.66E-09	1.39E-08	4.21E-09	2
Pm-147	1.69E-12	2.48E-12	4.88E-12	1.61E-11	3.21E-11	
Pu-238	1.31E-10	1.69E-10	2.72E-10	3.36E-08	7.22E-11	
Pu-239	1.35E-10	1.74E-10	2.76E-10	3.33E-08	2.00E-10	
Pu-240	1.35E-10	1.74E-10	2.77E-10	3.33E-08	6.98E-11	
Pu-241	1.76E-12	2.28E-12	3.29E-12	3.34E-10	4.11E-12	
Pu-242	1.28E-10	1.65E-10	2.63E-10	3.13E-08	6.25E-11	
Pu-244+D	1.44E-10	1.90E-10	3.14E-10	2.93E-08	1.51E-06	2
Ra-226+D	3.86E-10	5.15E-10	7.30E-10	1.16E-08	8.49E-06	2
Ra-228+D	1.04E-09	1.43E-09	2.29E-09	5.23E-09	4.53E-06	2
Ru-106+D	4.22E-11	6.11E-11	1.19E-10	1.02E-10	9.66E-07	2
Sb-125+D	5.13E-12	7.21E-12	1.32E-11	1.93E-11	1.81E-06	2
Sm-147	3.74E-11	4.77E-11	7.59E-11	6.88E-09	0	
Sm-151	5.55E-13	8.07E-13	1.59E-12	4.88E-12	3.60E-13	
Sr-90+D	7.40E-11	9.53E-11	1.44E-10	1.13E-10	1.96E-08	2
Tc-99	2.75E-12	4.00E-12	7.66E-12	1.41E-11	8.14E-11	
Th-228+D	3.00E-10	4.22E-10	8.09E-10	1.43E-07	7.76E-06	2
Th-229+D	5.28E-10	7.16E-10	1.29E-09	2.25E-07	1.17E-06	2
Th-230	9.10E-11	1.19E-10	2.02E-10	2.85E-08	8.19E-10	
Th-232	1.01E-10	1.33E-10	2.31E-10	4.33E-08	3.42E-10	
Tl-204	5.85E-12	8.25E-12	1.54E-11	2.45E-12	2.76E-09	
U-232	2.92E-10	3.85E-10	5.74E-10	1.95E-08	5.98E-10	
U-233	7.18E-11	9.69E-11	1.60E-10	1.16E-08	9.82E-10	
U-234	7.07E-11	9.55E-11	1.58E-10	1.14E-08	2.52E-10	
U-235+D	7.18E-11	9.76E-11	1.63E-11	1.01E-08	5.43E-07	2
U-236	6.70E-11	9.03E-11	1.49E-10	1.05E-08	1.25E-10	
U-238+D	8.71E-11	1.21E-10	2.10E-10	9.35E-09	1.14E-07	2

Radionuclide	Water Ingestion (risk/pCi)	Food Ingestion (risk/pCi)	Soil Ingestion (risk/pCi)	Inhalation (risk/pCi)	External Exposure (risk/yr per PCi/g soil)	Notes
Zn-65	1.17E-11	1.54E-11	2.45E-11	5.81E-12	2.81E-06	

Notes:

1. A curie (Ci), the customary unit of activity, is equal to 3.7 x 10^{10} nuclear transformations per second. 1 picocurie (pCi) = 10^{-12} Ci. If required, slope factors in Table D.1 can be converted into the International System (SI) units of becquerels (1 Bq = 1 nuclear transformation per second) by multiplying each inhalation, ingestion, or external exposure value by 27.03. Users can calculate cancer risks using slope factors expressed in either customary units or SI units with equivalent results, provided that they also use air, water, food and soil concentration values in the same system of units.

2. For each radionuclide listed, slope factors correspond to the risks per unit intake or exposure for that radionuclide only, except when marked with a "+D". In these cases, the risks from associated short-lived radioactive decay products (i.e., those decay products with radioactive half-lives less than or equal to 6 months) are also included, based on an assumption of secular equilibrium. These decay chains are identified in Table C.1 of Attachment C.

3. The inhalation slope factor listed represents inhalation of C-14 as a particulate. Alternative values for inhalation of C-14 as a gas are 3.36E-15 risk/pCi for carbon monoxide and 1.99E-14 risk/pCi for carbon dioxide.

4. The inhalation slope factor for H-3 represents inhalation of titiated water vapor, which is considered the most likely form in the environment. Alternative values of inhalation of H-3 include 1.99E-13 risk/pCi for particulates, 5.62E-18 risk/pCi for elemental hydrogen gas, and 1.28E-13 risk/pCi for organic forms. Similarly, the ingestion slope factor values for H-3 represent ingestion of tritiated water, which is considered the most likely form in the environment. Alternative values for ingestion of organically bound forms of H-3 in water, food, and soil are 1.12E-13 risk/pCi, 1.44E-13 risk/pCi, and 2.02E-13 risk/pCi, respectively.

5. The food ingestion slope factor for I-129 represents ingestion of milk. For ingestion of non-dairy foodstuffs, a lower value of 1.93E-10 risk/pCi ingested would apply. The inhalation slope factor for I-129 represents inhalation of particulates; alternative values for inhalation of I-129 vapor are 1.24E-10 for inhalation of methyl iodide and 1.60E-10 for inhalation of other compounds in vapor form.

Table 2.3 Radionuclide Drinking Water MCLs

Radionuclide	Current MCL[a, b] (pCi/L)	Proposed MCL (pCi/L)	Risk Base Limit (RBL)[e] (pCi/L)	Mass Equiv to MCL, Proposed MCL, or RBL (mg/L)
Ac-227			0.24	3.3E-12
Ag-108m			5.8	2.2E-10
Ag-110m	90			1.9E-11
Am-241	15			4.4E-09
Am-243	15			7.5E-08
Bi-207	200			4.4E-09
C-14	2,000			4.5E-07
Cd-109	600			2.3E-10
Ce-144	30			9.1E-12
Cl-36	700			2.1E-05
Cm-243	15			2.9E-10
Cm-244	15			1.9E-10
Cm-248	15			3.5E-06
Co-57	1,000			1.2E-10
Co-60	100			8.9E-11
Cs-134	80			6.2E-11
Cs-135	900			7.8E-04
Cs-137	200			2.3E-09
Eu-152	200			1.1E-09
Eu-154	60			2.3E-10
Eu-155	600			1.3E-09
Fe-55	2,000			8.3E-10
Gd-153	600			1.7E-10
H-3	20,000			2.1E-09
I-129	1			5.7E-06
K-40			1.9	2.7E-4
Mn-54	300			3.9E-11
Na-22	400			6.4E-11
Nb-94			6.1	3.3E-8
Ni-59	300			3.7E-06
Ni-63	50			8.5E-10
Np-237	15			2.1E-05
Pa-231	15			3.2E-07
Pb-210			0.054	7.1E-13
Pm-147	587			6.3E-10
Pu-238	15			8.8E-10

Table 2.3 Radionuclide Drinking Water MCLs

Radionuclide	Current MCL[a, b] (pCi/L)	Proposed MCL (pCi/L)	Risk Base Limit (RBL)[e] (pCi/L)	Mass Equiv to MCL, Proposed MCL, or RBL (mg/L)
Pu-239	15			2.4E-07
Pu-240	15			6.6E-08
Pu-241			27	2.6E-10
Pu-242	15			3.8E-06
Pu-244	15			8.5E-04
Ra-226	5[c]			5.1E-09
Ra-228	5[c]			1.8E-11
Ru-106	30			9.0E-12
Sb-125	300			2.9E-10
Sm-147	15			6.5E-01
Sm-151	1,000			3.8E-08
Sr-90	8			5.9E-11
Tc-99	900			5.3E-05
Th-228	15			1.8E-11
Th-229	15			7.1E-08
Th-230	15			7.4E-07
Th-232	15			1.4E-01
Tl-204	300			6.5E-10
U-232		20[d]		9.4E-10
U-232		(20 μg/l)[d]		2.0E-02
U-233		20[d]		2.1E-06
U-233		(20 μg/l)[d]		2.0E-02
U-234		20[d]		3.2E-06
U-234		(20 μg/l)[d]		2.0E-02
U-235		20[d]		9.3E-03
U-235		(20 μg/l)[d]		2.0E-02
U-236		20[d]		3.1E-04
U-236		(20 μg/l)[d]		2.0E-02
U-238		20[d]		6.0E-02
U-238		(20 μg/l)[d]		2.0E-02
Zn-65	300			3.6E-11

Notes:

[a] Existing MCL is 4 mrem/yr to the whole body or an organ, combined from all beta and photon emitters.

[b] Existing MCL is 15 pCi/L, with the concentration level combined for all alpha emitters, except radon and uranium.

[c] Existing MCL is 5 pCi/L combined for Ra-226 and Ra-228.

[d] Preferred EPA proposed MCL standard is 20 μg/l and 20 pCi/l for uranium, with EPA soliciting comments on options of 40 μg/l **and** 40 pCi/l, and 80 μg/l **and** 80 pCi/l. The preferred proposed MCL standard for uranium of 20 μg/l and 20 pCi/l is represented in this table.

[e] Risk Based Limits are calculated for 30-year exposure duration and 10^{-6} risk.

2.1.1 Additive Risk. For all pathways except migration to ground water, SSLs correspond to a 10^{-6} lifetime cancer risk level. While the pathways included in the analysis are considered to represent a residential setting, SSLs are not calculated for a specific scenario (i.e., SSLs are not summed over a set of pathways). EPA believes that setting a 10^{-6} risk level for individual radionuclides and pathways generally will lead to cumulative risks within the 10^{-4} to 10^{-6} range for the combinations of radionuclides typically found at Superfund sites.

2.1.2 Toxicokinetics of Radionuclides. The exposure assumptions used to develop SSLs are representative of a chronic exposure pathway. Exposures necessary to cause acute radiation effects are many orders of magnitude greater than those associated with SSLs. Therefore, it is extremely unlikely that even massive intakes of contaminated soil could cause such effects. It should be noted, however, that the slope factors used to calculate SSLs in this report are applicable to either chronic or acute exposure to a radionuclide. That is, the slope factor may be interpreted either as the average risk per unit exposure to members of a population exposed throughout life to a constant concentration of a radionuclide in soil, or as the average risk per unit exposure to members of a population acutely exposed to the radionuclide in soil.

2.2 Direct Ingestion of Soil

Calculation of SSLs for direct ingestion of soil is based on the methodology presented for residential land use in RAGS HHEM, Part B (U.S. EPA, 1991b). Briefly, this methodology backcalculates a soil concentration level from a target risk. A number of studies have shown that inadvertent ingestion of soil is common among children 6 years old and younger (Calabrese et al., 1989; Davis et al., 1990; Van Wijnen et al., 1990). Therefore, the approach uses an age-adjusted soil ingestion factor that takes into account the difference in daily soil ingestion rates and exposure duration for children from 1 to 6 years old and others from 7 to 31 years old. The higher intake rate of soil by children leads to a lower, or more conservative, risk-based concentration compared to an adult-only assumption. RAGS HHEM, Part B also uses this age-adjusted approach.

For radionuclides, both the magnitude and duration of exposure are important. Duration is critical because the toxicity criteria are based on "lifetime average daily dose." Therefore, the total intake, whether it be over 5 years or 50 years, is averaged over a lifetime of 70 years. To be protective of exposures to radionuclides in the residential setting, RAGS HHEM, Part B (U.S. EPA, 1991b) and EPA focus on exposures to individuals who may live in the same residence for a "high-end" period of time (e.g., 30 years). As mentioned above, exposure to soil is higher during childhood and decreases with age. Thus, Equation 1 uses the RAGS HHEM, Part B time-weighted average soil ingestion rate for children and adults; the derivation of this factor is shown in Equation 2.

Screening Level Equation for Ingestion of Radionuclides in Residential Soil
(Source: RAGS HHEM, Part B; U.S. EPA, 1991b)

$$SSL = \frac{TR}{SF_s * IR_s * 1E-3 * EF * ED} \tag{1}$$

Parameter/Definition (units)	Default
TR/target cancer risk (unitless)	10^{-6}
SF_s /soil ingestion slope factor $(pCi)^{-1}$	See Table 2.2
IR_s /soil ingestion rate (mg/d)	120 (age-averaged)
1E-3/conversion factor (g/mg)	--
EF/exposure frequency (d/yr)	350
ED/exposure duration (yr)	30

Source: RAGS HHEM, Part B; U.S. EPA, 1991b

Equation for Age-Adjusted Soil Ingestion Factor, IR_s

$$IR_s = \frac{IR_{soil/age1-6} * ED_{age1-6} + IR_{soil/age7-31} * ED_{age7-31}}{ED} \tag{2}$$

Parameter/Definition (units)	Default
IR_s /age-adjusted soil ingestion factor (mg/d)	120
$IR_{soil/age1-6}$ /ingestion rate of soil age 1-6 (mg/d)	200
ED_{age1-6} /exposure duration during ages 1-6 (yr)	6
$IR_{soil/age7-31}$ /ingestion rate of soil age 7-31 (mg/d)	100
$ED_{age7-31}$ /exposure duration during ages 7-31 (yr)	24
ED/exposure duration (yr)	30

Source: RAGS HHEM, Part B (U.S. EPA, 1991b).

Because of the impracticability of developing site-specific input parameters (e.g., soil ingestion rates) for direct soil ingestion, SSLs are calculated using the defaults listed in Equations 1 and 2. Appendix A lists these generic SSLs for direct ingestion of soil.

2.3 Inhalation of Fugitive Dusts

The models and assumptions used to calculate SSLs for the inhalation pathway are presented in Equations 3 and 4, along with the default parameter values used to calculate the generic SSLs presented in Appendix A. Particular attention is given to the emissions portion and the dispersion portion (Q/F) of the particulate emission factor (PEF) equation, which has been revised since originally presented in RAGS HHEM, Part B. The default PEF presented here is the same as that given in the SSG for chemicals, which allows for the presence of a family garden associated with the ingestion of homegrown produce pathway in a residential setting. The available radionuclide-specific human health benchmarks used in these equations are presented in Section 2.1.

2.3.1 Screening Level Equation for Inhalation of Fugitive Dusts. Equation 3 is used to calculate SSLs for the inhalation of fugitive dusts containing radionuclides.

Screening Level Equation for Inhalation of Radionuclides in Fugitive Dusts in Residential Soil

$$SSL = \frac{TR}{SF_i * IR_i * (\frac{1}{PEF}) * 1E{+}3 * EF * ED * [ET_o + (ET_i * DF_i)]} \qquad \textbf{(3)}$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10^{-6}	U.S. EPA, 1991b
SF_i /inhalation slope factor (pCi^{-1})	See Table 2.2	See Part 2.1
IR_i /inhalation rate (m^3/d)	20	U.S. EPA, 1991b
PEF/particulate emission factor (m^3/kg)	1.32E+09 (Equation 3)	U.S. EPA, 1991b
1E+3/conversion factor (g/kg)	--	--
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b
ET_o/exposure time fraction, outdoor (unitless)	0.073	U.S. EPA, 1997b
ET_i/exposure time fraction, indoor (unitless)	0.683	U.S. EPA, 1997b
DF_i/dilution factor for indoor inhalation, (unitless)	0.4	Alonza, 1979

To calculate inhalation SSLs, the particulate emission factor must be calculated. The derivation of PEF has been updated since RAGS HHEM, Part B was published and is discussed fully in Sections 2.3.3. The PEF equation can be broken into two separate models: models to estimate the emissions of dusts, and a dispersion model (reduced to the term Q/C) that simulates the dispersion of radionuclides in the atmosphere.

2.3.2 Dispersion Model. The box model in RAGS HHEM, Part B has been replaced with a Q/C term derived from a modeling exercise using meteorologic data from 29 locations across the United States.

The dispersion model used in the Part B guidance is based on the assumption that emissions into a hypothetical box will be distributed uniformly throughout the box. To arrive at the volume within the box, it is necessary to assign values to the length, width, and height of the box. The length (LS) was the length of a side of a contaminated site with a default value of 45 m; the width was based on the windspeed in the mixing zone (V) with a default value of 2.25 m (based on a windspeed of 2.25 m/s); and the height was the diffusion height (DH) with a default value of 2 m.

However, the assumptions and mathematical treatment of dispersion used in the box model may not be applicable to a broad range of site types and meteorology and do not utilize state-of-the-art techniques developed for regulatory dispersion modeling. EPA was very concerned about the defensibility of the box model and sought a more defensible dispersion model that could be used as a replacement to the Part B guidance and had the following characteristics:

- Dispersion modeling from a ground-level area source
- Onsite receptor
- A long-term/annual average exposure point concentration
- Algorithms for calculating the exposure point concentration for area sources of different sizes and shapes.

To identify such a model, EPA held discussions with the EPA Office of Air Quality Planning and Standards (OAQPS) concerning recent efforts to develop a new algorithm for estimating ambient air concentrations from low or ground-level, nonbuoyant sources of emissions. The new algorithm is incorporated into the Industrial Source Complex Model (ISC2) platform in both a short-term mode (AREA-ST) and a long-term mode (AREA-LT). Both models employ a double numerical integration over the source in the upwind and crosswind directions. Wind tunnel tests have shown that the new algorithm performs well with onsite and near-field receptors. In addition, subdivision of the source is not required for these receptors.

Because the new algorithm provides better concentration estimates for onsite and for near-field receptors, a revised dispersion analysis was performed (Appendix D; EQ, 1994). The AREA-ST model was run for 0.5-acre and 30-acre square sources with a full year of meteorologic data for 29 U.S locations selected to be representative of the national range of meteorologic conditions (EQ, 1993). Additional modeling runs were conducted to address a range of square area sources from 0.5 to 30 acres in size (Table 2.4). The Q/C values in Table 2.4 for 0.5- and 30-acre sources differ slightly from the values in Appendix D due to differences in rounding conventions used in the final model runs.

To calculate site-specific SSLs, select a Q/C value from Table 2.4 that best represents a site's size and meteorologic condition.

To develop a reasonably conservative default Q/C for calculating generic SSLs, a default site (Minneapolis, MN) was chosen that best approximated the 90th percentile of the 29 normalized concentrations (kg/m^3 per g/m^2-s). The inverse of this concentration results in a default Q/C value of 90.80 g/m^2-s per kg/m^3 for a 0.5-acre site.

Table 2.4 . Q/C Values by Source Area, City, and Climatic Zone

	Q/C (g/m^2-s per kg/m^3)					
	0.5 Acre	**1 Acre**	**2 Acre**	**5 Acre**	**10 Acre**	**30 Acre**
Zone I						
Seattle	82.72	72.62	64.38	55.66	50.09	42.86
Salem	73.44	64.42	57.09	49.33	44.37	37.94
Zone II						
Fresno	62.00	54.37	48.16	41.57	37.36	31.90
Los Angeles	68.81	60.24	53.30	45.93	41.24	35.15
San Francisco	89.51	78.51	69.55	60.03	53.95	46.03
Zone III						
Las Vegas	95.55	83.87	74.38	64.32	57.90	49.56
Phoenix	64.04	56.07	49.59	42.72	38.35	32.68
Albuquerque	84.18	73.82	65.40	56.47	50.77	43.37
Zone IV						
Boise	69.41	60.88	53.94	46.57	41.87	35.75
Winnemucca	69.23	60.67	53.72	46.35	41.65	35.55
Salt Lake City	78.09	68.47	60.66	52.37	47.08	40.20
Casper	100.13	87.87	77.91	67.34	60.59	51.80
Denver	75.59	66.27	58.68	50.64	45.52	38.87
Zone V						
Bismark	83.39	73.07	64.71	55.82	50.16	42.79
Minneapolis	90.80	79.68	70.64	61.03	54.90	46.92
Lincoln	81.64	71.47	63.22	54.47	48.89	41.65
Zone VI						
Little Rock	73.63	64.51	57.10	49.23	44.19	37.64
Houston	79.25	69.47	61.53	53.11	47.74	40.76
Atlanta	77.08	67.56	59.83	51.62	46.37	39.54
Charleston	74.89	65.65	58.13	50.17	45.08	38.48
Raleigh-Durham	77.26	67.75	60.01	51.78	46.51	39.64
Zone VII						
Chicago	97.78	85.81	76.08	65.75	59.16	50.60
Cleveland	83.22	73.06	64.78	55.99	50.38	43.08
Huntington	53.89	47.24	41.83	36.10	32.43	27.67
Harrisburg	81.90	71.87	63.72	55.07	49.56	42.40
Zone VIII						
Portland	74.23	65.01	57.52	49.57	44.49	37.88
Hartford	71.35	62.55	55.40	47.83	43.00	36.73
Philadelphia	90.24	79.14	70.14	60.59	54.50	46.59
Zone IX						
Miami	85.61	74.97	66.33	57.17	51.33	43.74

2.3.3 Particulate Emission Factor. The particulate emission factor relates the concentration of contaminant in soil with the concentration of dust particles in the air. This guidance addresses dust generated from open sources, which is termed "fugitive" because it is not discharged into the atmosphere in a confined flow stream. Other sources of fugitive dusts that may lead to higher emissions due to mechanical disturbances include unpaved roads, tilled agricultural soils, and heavy construction operations.

Both the emissions portion and the dispersion portion (Q/C) of the PEF equation have been updated since RAGS HHEM, Part B.

As in Part B, the emissions part of the PEF equation is based on the "unlimited reservoir" model from Cowherd et al. (1985) developed to estimate particulate emissions due to wind erosion. The unlimited reservoir model is most sensitive to the threshold friction velocity, which is a function of the mode of the size distribution of surface soil aggregates. This parameter has the greatest effect on the emissions and resulting concentration. For this reason, a conservative mode soil aggregate size of 500 µm was selected as the default value for calculating generic SSLs.

The mode soil aggregate size determines how much wind is needed before dust is generated at a site. A mode soil aggregate size of 500 µm yields an **uncorrected** threshold friction velocity of 0.5 m/s. This means that the windspeed must be at least 0.5 m/s before any fugitive dusts are generated. However, the threshold friction velocity **should be corrected** to account for the presence of nonerodible elements. In Cowherd et al. (1985), nonerodible elements are described as:

> . . . clumps of grass or stones (larger than about 1 cm in diameter) on the surface (that will) consume part of the shear stress of the wind which otherwise would be transferred to erodible soil.

Cowherd et al. describe a study by Marshall (1971) that used wind tunnel studies to quantify the increase in the threshold friction velocity for different kinds of nonerodible elements. His results are presented in Cowherd et al. as a graph showing the rate of corrected to uncorrected threshold friction velocity vs. L_c, where L_c is a measure of nonerodible elements vs. bare, loose soil. Thus, the ratio of corrected to uncorrected threshold friction velocity is directly related to the amount of nonerodible elements in surface soils.

Using a ratio of corrected to uncorrected threshold friction velocity of 1, or no correction, is roughly equivalent to modeling "coal dust on a concrete pad," whereas using a correction factor of 2 corresponds to a windspeed of 19 m/s at a height of 10 m. This means that about a 43-mph wind would be required to produce any particulate emissions. Given that the 29 meteorologic data sets used in this modeling effort showed few windspeeds at, or greater than, 19 m/s, EPA felt that it was necessary to choose a default correction ratio between 1 and 2. A value of 1.25 was selected as a reasonable number that would be at the more conservative end of the range. This equates to a corrected threshold friction velocity of 0.625 m/s and an equivalent windspeed of 11.3 m/s at a height of 7 meters. The default PEF is the same as for chemicals. Another, key assumption in the derivation of the PEF is that the ½ acre lot has only 50% vegetative cover. Although the ingestion of homegrown produce is not quantitatively evaluated in the SSG for chemicals, the assumption of 50% vegetative cover allows for the presence of a family garden.

Q/C values are needed to calculate the PEF (Equation 4); use the Q/C value in Table 2.4 that best represents a site's size and meteorologic conditions. Cowherd et al. (1985) describe how to obtain site-specific estimates of V, U_m, U_t, and F(x).

Meteorologic conditions (i.e., the intensity and frequency of wind) affect both the dispersion and emissions of particulate matter. For this reason, a separate default Q/C value was derived for particulate matter [nominally 10 µm and less (PM_{10})] emissions for the generic SSLs. The PEF equation was used to calculate annual average concentrations for each of 29 sites across the country. To develop a reasonably conservative default Q/C for calculating generic SSLs, a default site (Minneapolis, MN) was selected that best approximated the 90th percentile concentration.

The results produced a revised default PEF Q/C value of 90.80 g/m^2-s per kg/m^3 for a 0.5-acre site (see Appendix D; EQ, 1994). The generic PEF derived using the default values in Equation 4 is 1.32×10^9 m^3/kg, which corresponds to a receptor point concentration of approximately 0.76 $µg/m^3$. This represents an annual average emission rate based on wind erosion that should be compared with chronic health criteria; it is **not** appropriate for evaluating the potential for more acute exposures.

$$PEF = Q/C * \frac{3{,}600}{0.036 * (1-V) * (U_m/U_t)^3 * F(x)} \qquad \textbf{(4)}$$

Parameter/Definition (units)	Default	Source
PEF/particulate emission factor (m^3/kg)	1.32 x 10^9	- -
Q/C/inverse of mean conc. at center of square source (g/m^2-s per kg/m^3)	90.80	Table 3 (for 0.5-acre source in Minneapolis, MN)
V/fraction of vegetative cover (unitless)	0.5 (50%)	U.S. EPA, 1991b
U_m/mean annual windspeed (m/s)	4.69	EQ, 1994
U_t/equivalent threshold value of windspeed at 7 m (m/s)	11.32	U.S. EPA, 1991b
F(x)/function dependent on U_m/U_t derived using Cowherd et al. (1985) (unitless)	0.194	U.S. EPA, 1991b

2.4 External Exposure from Radionuclides in Soil

Individuals residing on a contaminated site will be exposed to photons emitted by those radionuclides present in the soil. In modeling external exposure to contaminated soil, the RAGS/HHEM Part B model (EPA91a) does not account for the following processes:

- radioactive decay and progeny ingrowth (i.e., radioactive daughters);
- correction factors for the geometry of the contaminated soil;
- depletion of the contaminated soil horizon by environmental processes, such as leaching, erosion, or plant uptake; and
- corrections for shielding by clean cover material.

The RAGS/HHEM Part B model effectively assumes that an individual is continually exposed to a non-depleting source term with a geometry that is effectively an infinite slab. The concept of an “infinite slab” means that the thickness of the contaminated zone and its aerial extent are so large that it behaves as if it were infinite in its physical dimensions. In practice, soil contaminated to a depth greater than about 15 cm and with an aerial extent greater than about 1,000 m^2 will create a radiation field comparable to that of an infinite slab. For calculation of SSLs for a residential setting, an adjustment for small areas is considered to be an important modification to the RAGS/HHEM Part B model, since in most residential settings the assumption of an infinite slab source will result in overly conservative SSLs. Thus, an area correction factor, ACF, has been added to the model for the calculation of SSLs. For the purposes of this report, adjustments for clean cover are not needed since, in all cases, it is assumed that the contaminated soil extends to the surface. The model provides adjustments for indoor occupancy and associated shielding effects by the simple application of a shielding factor and indoor occupancy time adjustment.

Screening Level Equation for External Exposure to Radionuclides in Soil

$$SSL = \frac{TR}{SF_e * \frac{EF}{365} * ED * ACF * [ET_o * (ET_i * GSF)]} \qquad \textbf{(5)}$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10^{-6}	U.S. EPA, 1991b
SF_e /external exposure slope factor (g/pCi/yr)	See Table 2.2	See Part 2.1
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b
ACF/area correction factor	0.9	ANL, 1993b
ET_o/exposure time fraction, outdoor (unitless)	0.073	U.S. EPA, 1997b
ET_i/exposure time fraction, indoor (unitless)	0.683	U.S. EPA, 1997b
GSF/gamma shielding factor	0.4	U.S. EPA, 1996d

With the exception of the area correction factor, default values are used for all input parameters in Equation 5 to calculate generic external exposure SSLs. The amount of data required to derive site-specific values for these parameters makes their collection and use impracticable for calculation of simple site-specific SSLs. Therefore, site-specific data are generally not available for this exposure pathway. An area correction factor less than 0.9 will only apply to very small sites (i.e., those with an area less than 1,000 m^2). Alternative area correction factors are discussed in Section 5.1.

2.4.1. Gamma Shielding Factor - GSF (unitless)

The gamma shielding factor is the ratio of the external gamma radiation level indoors on site to the radiation level outdoors on-site. It is based on the fact that a building provides shielding against penetration of gamma radiation. Therefore, the calculation of the risk posed by gamma radiation from radionuclides in the soil should take into account this shielding effect.

EPA's previous gamma shielding factor default value, taken from RAGS/HHEM Part B (U.S. EPA, 1991b), is 0.8, which assumes that the external gamma radiation level indoors is 20% lower than the outdoor gamma radiation level. This value is based on information presented in two EPA reports, *Natural Radiation in the United States* (U.S. EPA, 1972) and *Population Exposure to External Natural Radiation Background in the United States* (U.S. EPA, 1981). Based on a review of the literature provided in these reports, EPA concluded that external background exposures in frame dwellings are 70 to 80% of outdoor values. This conclusion is based on empirical data for natural background radiation. As such, it includes the contribution from both terrestrial and cosmic radiation and from the radionuclides present in structure material.

A further review of the literature performed in the EPA report, *Reassessment of Radium and Thorium Soil Concentrations and Annual Dose Rates* (U.S. EPA, 1996d) reveals numerous publications that address indoor/outdoor gamma ray shielding factors as applied to radioactive fallout from nuclear weapons and reactor accidents. In U.S. EPA, 1981, the authors performed a review of experimentally measured reduction factors from fallout. The authors concluded that "reduction factors of 0.4 to 0.2 are recommended as representative values for above-ground lightly constructed (wood frame) and heavily constructed (block and brick) homes, respectively." On the basis of this review, U.S. EPA 1996d, suggests that a default gamma shielding factor of 0.4 based solely on the contribution of terrestrial radiation might be a more appropriate value to use at sites with soil contaminated with radionuclides than the previous EPA default of 0.8 which also included the effects of cosmic radiation and the inherent radioactivity in structure materials. Based on this rationale, the value of 0.4 is adopted in this guidance as EPA's new default gamma shielding factor.

2.5 Ingestion of Homegrown Produce.

Persons living on a contaminated site may ingest radioactive material by consumption of plants grown in a family garden. In this model, the fruits and vegetables primarily become contaminated by root uptake of radionuclides contained in the pore water of the soil in which the plants are growing.

Residential gardens that provide a significant fraction of the family diet are not uncommon, particularly in more rural areas. However, it is quite uncommon that such a family garden would be totally self-sufficient. To be self-sufficient, the family would have to raise all of its vegetables onsite—that is, grow its own potatoes, get all fruit from an onsite orchard, can or freeze enough of the summer crop to provide all food during the winter, and raise all grain used for bread and cereal. The model uses available consumption data for home grown produce from USDA surveys.

The model accounts for root uptake with a simple soil-to-plant transfer factor. These soil-to-plant transfer factors have been developed based upon the assumption that the entire plant root system is wholly exposed to contaminated soil. If the plant roots extend to a depth of 100 cm but the radionuclide contaminants are confined to the upper 15 cm, an initial assumption may be that only 15% of the root system is active in accumulating contaminants and that the reported soil-to-plant transfer factors should be reduced by a correction factor of 0.15. However, the equation for calculation of SSLs for this pathway does not apply any reduction to the soil-to-plant transfer factors. The basis for this assumption is as follows.

Most plant root systems are in fact very active in the upper soil horizon, especially in the upper 15 cm of soil. This point is illustrated in a number of ways: 1) by illustrations of root morphology and growth habit, 2) positive physiological factors including the availability of water, oxygen and nutrients near the soil surface, 3) negative physiological or agronomic factors—including subsurface soil compaction, subsurface zones of acidity, perched water tables, hypoxia, etc., 4) interactions with soil microbes—with a special focus on mychorrizal fungi, and 5) split root studies. Thus, roots commonly proliferate in the upper layers of soil. If one assumes that a plant is actively growing, then ion uptake characteristics and lateral root growth strongly suggest that simply attributing 15% of root uptake activity to the upper 15 cm of the soil is not a sound approach. Environmental forces may influence root growth to one or more meters in depth, but more so for obtaining water than nutrients. In reality, the upper 15 cm of soil may include 50% or more of the root system—and thus 50% or more of the ion uptake (SC&A, 1994).

The decision to not include air deposition or rain-splash does not affect any radionuclides because the increase in concentration from this route is not significant or is markedly reduced when peeling, washing, cooking, and other food preparation processes are taken into consideration (U.S. EPA, 1994k). The decision to not include the irrigation pathway is only an issue when there is medium to heavy irrigation using contaminated water for a radionuclide with a long half-life, and an insignificant contribution from external exposure. The model also makes a conservative assumption to ignore the decay between harvest and ingestion and any removal during food processing.

The model does not include any special calculations for estimating concentrations of ^{3}H and ^{14}C in plants. Such calculations assume that a state of equilibrium exists among the concentrations of ^{3}H and ^{14}C in all environmental media—air, water, food products, and body tissues. This assumption may be conservative for a radioactively contaminated site with a finite area, but may be appropriate for an individual pathway, such as soil-to-plant pathway (ANL93a). For these calculations, the ^{3}H concentration in the plant is assumed to be the same as that in the contaminated water to which the plant is exposed. Similarly, the specific activity of ^{14}C in the plant (i.e., pCi/g of ^{14}C per gram of carbon in the plant) is the same as that of the ambient CO_2.

Other models use a factor, known as the contaminated plant fraction (CPF), to adjust for the fraction of fruits and vegetables obtained from the contaminated site relative to the total consumption rate. The ingestion rate used in these models is a total ingestion rate, which, when multiplied by the contaminated plant fraction, gives the ingestion rate of contaminated fruits and vegetables. However, the calculation of SSLs for this pathway makes use of published estimates of individual home produced fruit and vegetable intake, thus eliminating the need for a CPF.

Default values are used for all input parameters in Equation 5 to calculate SSLs for this pathway. The amount of data required to derive site-specific values for these parameters makes their collection and use impracticable for calculation of simple site-specific SSLs. Therefore, site-specific data are generally not available for this exposure pathway. The generic SSLs presented in the TBD are recommended for all sites.

Screening Level Equation for Ingestion of Radionuclides in Homegrown Produce

$$SSL = \frac{TR}{SF_p * (IR_f + IR_v) * TF_p * EF * ED} \qquad \textbf{(6)}$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10^{-6}	U.S. EPA, 1991b
SF_p /produce ingestion slope factor $(pCi)^{-1}$	See Table 2.2	See Part 2.1
IR_f /home grown fruits ingestion rate (g/d)	189	U.S. EPA, 1997b
IR_{lv} /home grown vegetables ingestion rate (g/d)	147	U.S. EPA, 1997b
TF_p/soil-to-plant transfer factor (pCi/g plant per pCi/g soil)	See Part 3.5.5	U.S. EPA, 1997b
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b

2.6 Migration to Ground Water

The methodology for calculating SSLs for the migration to ground water pathway was developed to identify radionuclide concentrations in soil that have the potential to contaminate ground water. Migration of radionuclides from soil to ground water can be envisioned as a two-stage process: (1) release of contaminant in soil leachate and (2) transport of the contaminant through the underlying soil and aquifer to a receptor well. The SSL methodology considers both of these fate and transport mechanisms.

The methodology incorporates a standard linear equilibrium soil/water partition equation to estimate radionuclide release in soil leachate (see Sections 2.6.1 through 2.6.4) and a simple water-balance equation that calculates a dilution factor to account for dilution of soil leachate in an aquifer (see Section 2.6.5). The dilution factor represents the reduction in soil leachate radionuclide concentrations by mixing in the aquifer, expressed as the ratio of leachate concentration to the concentration in ground water at the receptor point (i.e., drinking water well). Because the infinite source (i.e., steady-state) assumption can result in mass-balance violations for soluble contaminants and small sources, mass-limit models are provided that limit the amount of contaminant migrating from soil to ground water to the total amount of contaminant present in the source (see Section 2.7).

SSLs are backcalculated from acceptable ground water concentrations (i.e., MCLs, see Section 2.1). First, the acceptable ground water concentration is multiplied by a dilution factor to obtain a target leachate concentration. For example, if the dilution factor is 10 and the acceptable ground water concentration is 10 pCi/L, the target soil leachate concentration would be 100 pCi/L. The partition equation is then used to calculate the total soil concentration (i.e., SSL) corresponding to this soil leachate concentration.

The methodology for calculating SSLs for the migration to ground water pathway was developed under the following constraints:

- Because of the large nationwide variability in ground water vulnerability, the methodology should be flexible, allowing adjustments for site-specific conditions if adequate information is available.

- To be appropriate for early-stage application, the methodology needs to be simple, requiring a minimum of site-specific data.

- The methodology should be consistent with current understanding of subsurface processes.

- The process of developing and applying SSLs should generate information that can be used and built upon as a site evaluation progresses.

Flexibility is achieved by using readily obtainable site-specific data in standardized equations; conservative default input parameters are also provided for use when site-specific data are not available. In addition, more complex unsaturated zone fate-and-transport models have been identified that can be used to calculate SSLs when more detailed site-specific information is available or can be obtained (see Part 3). These models can extend the applicability of SSLs to subsurface conditions that are not adequately addressed by the simple equations (e.g., deep water tables; clay layers or other unsaturated zone characteristics that can attenuate contaminants before they reach ground water).

The SSL methodology was designed for use during the early stages of a site evaluation when information about subsurface conditions may be limited. Because of this constraint, the methodology is based on conservative, simplifying assumptions about the release and transport of contaminants in the subsurface (see Highlight 2).

Highlight 2: Simplifying Assumptions for the Migration to Ground Water Pathway

- The source is infinite (i.e., steady-state concentrations will be maintained in ground water over the exposure period of interest).
- Contaminants are uniformly distributed throughout the zone of contamination.
- Soil contamination extends from the surface to the water table (i.e., adsorption sites are filled in the unsaturated zone beneath the area of contamination).
- There is no chemical or biological degradation in the unsaturated zone.
- Equations in this document do not account for decay, however electronic version of these equations will account for decay in the unsaturated zone.
- Equilibrium soil/water partitioning is instantaneous and linear in the contaminated soil.
- The receptor well is at the edge of the source (i.e., there is no dilution from recharge downgradient of the site) and is screened within the plume.
- The aquifer is unconsolidated and unconfined (surficial).
- Aquifer properties are homogeneous and isotropic.
- Chelating or complexing agents not present.
- No facilitated transport (e.g., colloidal transport) of inorganic contaminants in aquifer.

Although simplified, the SSL methodology described in this section is theoretically and operationally consistent with the more sophisticated investigation and modeling efforts that are conducted to develop soil cleanup goals and cleanup levels for protection of ground water at Superfund sites. SSLs developed using this methodology can be viewed as evolving risk-based levels that can be refined as more site information becomes available. The early use of the methodology at a site will help focus further subsurface investigations on areas of true concern with respect to ground water quality and will provide information on soil characteristics, aquifer characteristics, and radionuclide properties that can be built upon as a site evaluation progresses.

2.6.1 Development of Soil/Water Partition Equation. The methodology used to estimate radionuclide release in soil leachate is based on the Freundlich equation, which was developed to model sorption from liquids to solids. The basic Freundlich equation applied to the soil/water system is:

$$K_d = C_s / C_w^n \tag{7}$$

where

K_d = Freundlich soil/water partition coefficient (L/kg)
C_s = concentration sorbed on soil (mg/kg)
C_w = soil leachate concentration (mg/L)
n = Freundlich exponent (dimensionless).

Assuming that adsorption is linear with respect to concentration (n=1) and rearranging to backcalculate a sorbed concentration (C_s):

$$C_s = (K_d)\ C_w \tag{8}$$

For SSL calculation, C_l is the target soil leachate concentration.

Adjusting Sorbed Soil Concentrations to Total Concentrations. To develop a screening level for comparison with contaminated soil samples, the sorbed concentration derived above (C_s) must be related to the total concentration measured in a soil sample (C_t). In a soil sample, contaminants can be associated with the solid soil materials or the soil water, and the soil air as follows (Feenstra et al., 1991):

$$M_t = M_s + M_w + M_a \tag{9}$$

where

M_t = total contaminant mass in sample (mg)
M_s = contaminant mass sorbed on soil materials (mg)
M_w = contaminant mass in soil water (mg)
M_a = contaminant mass in soil air (mg).

Furthermore,

$$M_t = C_t\ \rho_b\ V_{sp}\ , \tag{10}$$

$$M_s = C_s\ \rho_b\ V_{sp}\ , \tag{11}$$

$$M_w = C_w\ \theta_w\ V_{sp}\ , \tag{12}$$

and

$$M_a = C_a\ \theta_a\ V_{sp}\ , \tag{13}$$

where

ρ_b = dry soil bulk density (kg/L)
V_{sp} = sample volume (L)
θ_w = water-filled porosity (L_{water}/L_{soil})
C_a = concentration on soil pore air (mg/L_{soil})
θ_a = air-filled soil porosity (L_{air}/L_{soil}).

For contaminated soils (with concentrations below C_{sat}), C_a may be determined from C_w and the dimensionless Henry's law constant (H') using the following relationship:

$$C_a = C_w H^1 \qquad \textbf{(14)}$$

thus

$$M_a = C_w H^l \theta_a V_{sp} \qquad \textbf{(15)}$$

Substituting into Equation 9:

$$C_t = \frac{C_s \rho_b + C_w \theta_w + C_w H' \theta_a}{\rho_b} \qquad \textbf{(16)}$$

or

$$C_s = C_t - C_w \left(\frac{\theta_w + \theta_a H'}{\rho_b} \right) \qquad \textbf{(17)}$$

Substituting into Equation 8 and rearranging:

$$C_t = C_w * \left(K_d + \frac{\theta_w + \theta_a * H'}{\rho_b} \right) \qquad \textbf{(18)}$$

For most radionuclides there is no significant vapor pressure and the dimensionless Henry's Law constant (H^l) may be assumed to be zero, (except for radon). When H' is zero, Equation 18 becomes:

Soil-Water Partition Equation for Migration to Ground Water: Inorganic Contaminants:

$$SSL = C_t = C_w \left(K_d + \frac{\theta_w}{\rho_b} \right) \qquad \textbf{(19)}$$

Parameter/Definition (units)	Default	Source
SSL /screening level in soil (mg/kg)	--	--
C_w/target soil leachate concentration (mg/L)	(MCL) × 20 DAF	Table 2.3 (MCL); Section 2.6.2 (DAF for 0.5-acre source)
K_d/soil-water partition coefficient (L/kg)	radionuclide -specific	see Section 3.3.1
θ_w/water-filled soil porosity (L_{water}/L_{soil})	0.3 (30%)	U.S. EPA/ORD
n/total soil porosity (L_{pore}/L_{soil})	0.43	1 - ρ_b / ρ_s
ρ_b/dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s/soil particle density (kg/L)	2.65	U.S. EPA, 1991b

Since MCLs for radionuclides are given in radiological activity units of picocuries per liter of water (pCi/l), equation 19 can also be written as:

$$SSL = C_t = C_w (1 * 10^{-3}) \left(K_d + \frac{\theta_w}{\rho_b} \right) \qquad \textbf{(20)}$$

Parameter/Definition (units)	Default	Source
SSL /screening level in soil (pCi/g)	--	--
C_w/target soil leachate concentration (pCi/L)	(MCL) × 20 DAF	Table 2.3 (MCL); Section 2.6.2 (DAF for 0.5-acre source)
1×10^{-3}/conversion factor (kg/g)	--	--
K_d/soil-water partition coefficient (L/kg)	radionuclide -specific	see Section 3.3.1
θ_w/water-filled soil porosity (L_{water}/L_{soil})	0.3 (30%)	U.S. EPA/ORD
n/total soil porosity (L_{pore}/L_{soil})	0.43	$1 - \rho_b/\rho_s$
ρ_b/dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s/soil particle density (kg/L)	2.65	U.S. EPA, 1991b

Either Equation 19 or Equation 20 can be used to calculate SSLs (total soil concentrations, C_t) corresponding to soil leachate concentrations (C_w) equal to the target contaminant soil leachate concentration. The equation assumes that soil, water and solids are conserved during sampling.

Defaults. The User's Guide (U.S. EPA, 1996a) describes how to develop site-specific estimates of the soil parameters needed to calculate SSLs.

Sensitivity analyses have shown that soil bulk density (ρ_b) has too limited a range for surface soils (generally between 1.3 and 1.7 g/cm^3) to affect results with nearly the significance of soil moisture conditions. Therefore, a default bulk density of 1.50 g/cm^3, the mode of the range given for U.S. soils in the *Superfund Exposure Assessment Manual* (U.S. EPA, 1988), was chosen to calculate generic SSLs. This value is also consistent with the mean porosity (0.43) for loam soil presented in Carsel and Parrish (1988). The default value of ρ_s (2.65 g/cm^3) was taken from U.S. EPA (1988) as the particle density for most soil mineral material.

Because migration to ground water SSLs are not particularly sensitive to soil water content (see Section 2.5.7), a value for average water-filled soil porosity that is more typical of subsurface conditions (0.30) was used. This value is between the mean field capacity (0.20) of Class B soils (Carsel et al., 1988) and the saturated volumetric water content for loam (0.43).

K_d varies by radioactive element and soil type, as discussed below.

2.6.2 Inorganics (Metals, including radionuclides)—Partition Theory. Equations 19 or 20 can be used to estimate SSLs for radionuclides for the migration to ground water pathway. The derivation of K_d values is much more complicated for metals, including radionuclides, than for organic compounds. Unlike organic compounds, for which K_d values are largely controlled by a single parameter (soil organic carbon), K_d values for metals are significantly affected by a variety of soil conditions. The most significant parameters are pH, oxidation-reduction conditions, iron oxide content, soil organic matter content, cation exchange capacity, and major ion chemistry. The number of significant influencing parameters, their variability in the field, and differences in experimental methods result in a wide range of K_d values for individual metals reported in the literature (over several orders of magnitude). Thus, it is much more difficult to derive generic K_d values for metals than for organics. For a more detailed discussion of the K_d model concept, see *Understanding Variation in Partition Coefficient, K_d Values, Volume I & Volume II* (U.S. EPA , 1999b).

2.6.3 Assumptions for Soil/Water Partition Theory. The following assumptions are implicit in the SSL partitioning methodology. *These assumptions and their implications for SSL accuracy should be read and understood before using this methodology to calculate SSLs.*

1. **There is no contaminant loss due to volatilization or degradation.** The source is considered to be infinite; i.e., these processes do not reduce soil leachate concentrations over time. This is a **conservative** assumption, especially for smaller sites.

2. **Adsorption is linear with concentration.** The methodology assumes that adsorption is independent of concentration (i.e., the Freundlich exponent = 1). This assumption is valid at low concentrations (e.g., at levels close to the MCL) for most chemicals. As concentrations increase, however, the adsorption isotherm can depart from the linear.

3. **The system is at equilibrium with respect to adsorption.** This ignores adsorption/desorption kinetics by assuming that the soil and pore water concentrations are at equilibrium levels. In other words, the pore-water residence time is assumed to be longer than the time it takes for the system to reach equilibrium conditions.

   This assumption is **conservative**. If equilibrium conditions are not met, the concentration in the pore water will be less than that predicted by the methodology. The kinetics of adsorption are not adequately understood for a sufficient number of chemicals and site conditions to consider equilibrium kinetics in the methodology.

4. **Adsorption is reversible.** The methodology assumes that desorption processes operate in the same way as adsorption processes, since most of the partition coefficients are measured by adsorption experiments rather than by desorption experiments. In actuality, desorption is slower to some degree than adsorption and, in some cases, radionuclides can be irreversibly bound to the soil matrix. In general, the significance of this effect increases with increasing K_d. Note, adsorption is only one of several chemical mechanisms that may reduce the aqueous concentrations in soil or ground water. For a detailed discussion of these chemical mechanisms, see U.S. EPA, 1999b.

   This assumption is **conservative**. Slower desorption rates and irreversible sorption will result in lower pore-water concentrations than that predicted by the methodology. Again, the level of knowledge on desorption processes is not sufficient to consider desorption kinetics and degree of reversibility for all of the subject radionuclides.

2.6.4 Dilution/Attenuation Factor Development. As radionuclides in soil leachate move through soil and ground water, they are subjected to physical, chemical, and biological processes that tend to reduce the eventual contaminant concentration at the receptor point (i.e., drinking water well). These processes include adsorption onto soil and aquifer media, chemical transformation (e.g., hydrolysis, precipitation), biological degradation, and dilution due to mixing of the leachate with ambient ground water. The reduction in concentration can be expressed succinctly by a DAF, which is defined as the ratio of radionuclides concentration in soil leachate to the concentration in ground water at the receptor point. When calculating SSLs, a DAF is used to backcalculate the target soil leachate concentration from an acceptable ground water concentration (e.g., MCL). For example, if the acceptable ground water concentration is 10 pCi/L and the DAF is 10, the target leachate concentration would be 100 pCi/L.

The SSL methodology addresses only one of these dilution-attenuation processes: radionuclide dilution in ground water. A simple equation derived from a geohydrologic water-balance relationship has been developed for the methodology, as described in the following subsection. The ratio factor calculated by this equation is referred to as a dilution factor rather than a DAF because it does not consider processes that attenuate radionuclides in the subsurface (i.e., adsorption and degradation processes). This simplifying assumption was necessary for several reasons.

First, the infinite source assumption (constant source concentrations maintained over period of interest) results in all subsurface adsorption sites being eventually filled and no longer available to attenuate radionuclides. Second, soil contamination extends to the water table, eliminating attenuation processes in the unsaturated zone. Additionally, the receptor well is assumed to be at the edge of the source, minimizing the opportunity for attenuation in the aquifer. Finally, chemical-specific biological and chemical degradation rates are not known for many of the SSL chemicals;

where they are available they are usually based on laboratory studies under simplified, controlled conditions. Because natural subsurface conditions such as pH, redox conditions, soil mineralogy, and available nutrients have been shown to markedly affect natural chemical and biological degradation rates, and because the national variability in these properties is significant and has not been characterized, EPA does not believe that it is possible at this time to incorporate these degradation processes into the simple site-specific methodology for national application.

If adsorption or degradation processes are expected to significantly attenuate radionuclide concentrations at a site (e.g., for sites with deep water tables or soil conditions that will attenuate radionuclides), the site manager is encouraged to consider the option of using more sophisticated fate and transport models. Many of these models can consider adsorption and degradation processes and can model transient conditions necessary to consider a finite source size. Part 3 of this document presents information on the selection and use of such models for SSL application.

The dilution factor model assumes that the aquifer is unconfined and unconsolidated and has homogeneous and isotropic properties. Unconfined (surficial) aquifers are common across the country, are vulnerable to contamination, and can be used as drinking water sources by local residents. Dilution model results may not be applicable to fractured rock or karst aquifer types. The site manager should consider use of more appropriate models to calculate a dilution factor (or DAF) for such settings.

In addition, the simple dilution model does not consider facilitated transport. This ignores processes such as colloidal transport, transport via solvents other than water (e.g., NAPLs), and transport via dissolved organic matter (DOM). However, the transport via solvents other than water is operative only if certain site-specific conditions are present. Transport by DOM and colloids has been shown to be potentially significant under certain conditions in laboratory and field studies. Although much research is in progress on these processes, the current state of knowledge is not adequate to allow for their consideration in SSL calculations.

If there is the potential for the presence of NAPLs in soils at the site or site area in question, SSLs should not be used for this area (i.e., further investigation is required). If NAPLs are suspected in site soils, refer to U.S. EPA (1992c) for additional guidance on how to estimate the potential for DNAPL occurrence in the subsurface.

Dilution Model Development. EPA evaluated four simple water balance models to adjust SSLs for dilution in the aquifer. Although written in different terms, all four options reviewed can be expressed as the same simple water balance equation to calculate a dilution factor, as follows:

Option 1 (ASTM):

$$\text{dilution factor} = (1 + U_{gw}\, d/IL) \tag{21}$$

where

U_{gw} = Darcy ground water velocity (m/yr)
d = mixing zone depth (m)
I = infiltration rate (m/yr)
L = length of source parallel to flow (m).

For Darcy velocity:

$$U_{gw} = Ki \tag{22}$$

where

K = aquifer hydraulic conductivity (m/yr)
i = hydraulic gradient (m/m).

Thus

$$\text{dilution factor} = 1 + (Kid/IL) \tag{23}$$

Option 2 (EPA Ground Water Forum):

$$\text{dilution factor} = (Q_p + Q_A)/Q_p \tag{24}$$

where

Q_p = percolation flow rate (m^3/yr)
Q_A = aquifer flow rate (m^3/yr)

For percolation flow rate:

$$Q_p = IA \tag{25}$$

where
A = facility area (m^2) = WL.

For aquifer flow rate:

$$Q_A = WdKi \tag{26}$$

where
W = width of source perpendicular to flow (m)
d = mixing zone depth (m).

Thus

$$\text{dilution factor} = (IA + WdKi)/IWL$$

$$\text{dilution factor} = 1 + (Kid/IL) \tag{27}$$

Option 3 (Summers Model):

$$C_w = (Q_p C_p)/(Q_p + Q_A) \tag{28}$$

where
C_w = ground water contaminant concentration (mg/L)
C_p = soil leachate concentration (mg/L)

given that

$$C_w = C_p/\text{dilution factor}$$

$$1/\text{dilution factor} = Q_p/(Q_p + Q_A)$$

or

$$\text{dilution factor} = (Q_p + Q_A)/Q_p \qquad \text{(see **Option 2**)}$$

Option 4 (EPA ORD/RSKERL):

$$\text{dilution factor} = (Q_p + Q_A)/Q_p = RX/RL \tag{29}$$

where

R = recharge rate (m/yr) = infiltration rate (I, m/yr)
X = distance from receptor well to ground water divide (m)

(Note that the intermediate equation is the same as **Option 2**.)

This option is a longer-term option that is not considered further in this analysis because valid X values are not currently available either nationally or for specific sites. EPA is considering developing regional estimates for these parameters.

Dilution Model Input Parameters. As shown, all three options for calculating contaminant dilution in ground water can be expressed as the same equation (equation 30):

Ground Water Dilution Factor

$$\text{dilution factor} = 1 + (Kid/IL) \tag{30}$$

Parameter/Definition (units)
K/aquifer hydraulic conductivity (m/yr)
i/hydraulic gradient (m/m)
d/mixing zone depth (m)
I/infiltration rate (m/yr)
L/source length parallel to ground water flow (m)

Mixing Zone Depth (**d**). Because of its dependence on the other variables, mixing zone depth is estimated with the method used for the MULTIMED model (Sharp-Hansen et al., 1990). The MULTIMED estimation method was selected to be consistent with that used by EPA's Office of Solid Waste for the EPA Composite Model for Landfills (EPACML). The equation for estimating mixing zone depth (d) is as follows:

$$d = (2\ \alpha_v\ L)^{0.5} + d_a\ \{1 - \exp[(-\ L\ I)/(V_s\ n_e\ d_a)]\} \tag{31}$$

where

α_v = vertical dispersivity (m/m)
V_s = horizontal seepage velocity (m/yr)
n_e = effective aquifer porosity ($L_{pore}/L_{aquifer}$)
d_a = aquifer depth (m).

The first term, $(2\alpha_v L)^{0.5}$, estimates the depth of mixing due to vertical dispersivity ($d_{\alpha v}$) along the length of ground water travel. Defining the point of compliance with ground water standards at the downgradient edge of the source, this travel distance becomes the length of the source parallel to flow L. Vertical dispersivity can be estimated by the following relationship (Gelhar and Axness, 1981):

$$\alpha_v = 0.056\,\alpha_L \qquad \textbf{(32)}$$

where

α_L = longitudinal dispersivity = 0.1 x_r
x_r = horizontal distance to receptor (m).

Because the potential receptor is assumed to have a well at the edge of the facility, x_r = L and

$$\alpha_v = 0.0056\,L \qquad \textbf{(33)}$$

Thus

$$d_{\alpha v} = (0.0112\,L^2)^{0.5} \qquad \textbf{(34)}$$

The second term, d_a {1 - exp[(-LI) / ($V_s n_e d_a$)]}, estimates the depth of mixing due to the downward velocity of infiltrating water, d_{Iv}. In this equation, the following substitution may be made:

$$V_s = Ki/n_e \qquad \textbf{(35)}$$

so

$$d_{Iv} = d_a\,\{1 - \exp[(-\,L\,I)/(K\,I\,d_a)]\} \qquad \textbf{(36)}$$

Thus, mixing zone depth is calculated as follows:

$$d = d_{\alpha v} + d_{Iv} \qquad \textbf{(37)}$$

Estimation of Mixing Zone Depth

$$d = (0.0112\ L^2)^{0.5} + d_a\ \{1 - \exp[(-\ L\ I)/(K\ i\ d_a)]\} \quad \textbf{(38)}$$

Parameter/Definition (units)
d/mixing zone depth (m)
L/source length parallel to ground water flow (m)
I/infiltration rate (m/yr)
K/aquifer hydraulic conductivity (m/yr)
i/hydraulic gradient (m/m)
d_a/aquifer thickness (m)

Incorporation of this equation for mixing zone depth into the SSL dilution equation results in five parameters that must be estimated to calculate dilution: source length (L), infiltration rate (I), aquifer hydraulic conductivity (K), aquifer hydraulic gradient (i), and aquifer thickness (d_a). Aquifer thickness also serves as a limit for mixing zone depth. The User's Guide for Radionuclides (U.S. EPA, 1999a) describes how to develop site-specific estimates for these parameters. Parameter definitions and defaults used to develop generic SSLs are as follows:

- **Source Length (L)** is the length of the source (i.e., area of contaminated soil) parallel to ground water flow and affects the flux of contaminant released in soil leachate (IL) as well as the depth of mixing in the aquifer. The default option for this parameter assumes a square, 0.5-acre contaminant source. This default was changed from 30 acres in response to comments to be more representative of actual contaminated soil sources (see Section 1.3.4). Increasing source area (and thereby area) may result in a lower dilution factor. Appendix A includes an analysis of the conservatism associated with the 0.5-acre source size.

- **Infiltration Rate (I)**. Infiltration rate times the source area determines the amount of contaminant (in soil leachate) that enters the aquifer over time. Thus, increasing infiltration decreases the dilution factor. Two options can be used to generate infiltration rate estimates for SSL calculation. The first assumes that infiltration rate is equivalent to recharge. This is generally true for uncontrolled contaminated soil sites but would be conservative for capped sites (infiltration < recharge) and nonconservative for sites with an additional source of infiltration, such as surface impoundments (infiltration > recharge). Recharge estimates for this option can be obtained from Aller et al. (1987) by hydrogeologic setting, as described in Section 2.5.6. of the 1996 TBD (EPA 1996b). Recent additional information may also be found in "Estimation of Infiltration Rate in the Vadose Zone: Compilation of Simple Mathematical Models. Volume I" (U.S. EPA 1998c)

  The second option is to use the HELP model to estimate infiltration, as was done for OSW's EPACML and EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) modeling efforts. The Soil Screening Guidance (U.S. EPA, 1995c) provides information on obtaining and using the HELP model to estimate site-specific infiltration rates.

- **Aquifer Parameters**. Aquifer parameters needed for the dilution factor model include hydraulic conductivity (K, m/yr), hydraulic gradient (i, m/m), and aquifer thickness (d_a, m). The User's Guide (U.S. EPA, 1996a) describes how to develop aquifer parameter estimates for calculating a site-specific dilution factor.

2.6.5 Default Dilution-Attenuation Factor. EPA has selected a default DAF of 20 to account for contaminant dilution and attenuation during transport through the saturated zone to a compliance point (i.e., receptor well). This assumption was developed in Section 2.5.6 of the 1996 TBD (EPA 1996b). At most sites, this adjustment will more accurately reflect a radionuclide's threat to ground water resources than assuming a DAF of 1 (i.e., no dilution or attenuation). EPA selected a DAF of 20 using a "weight of evidence" approach. This approach considers results from OSW's EPACMTP model as well as results from applying the SSL dilution model described in Section 2.6.4 of this report to 300 ground water sites across the country.

The default DAF of 20 represents an adjustment from the DAF of 10 presented in the December 1994 draft Soil Screening Guidance (U.S. EPA, 1994h) to reflect a change in default source size from 30 acres to 0.5 acre. A DAF of 20 is protective for sources up to 0.5 acre in size. Analyses presented in Appendix A indicate that it can be protective of larger sources as well. However, this hypothesis should be examined on a case-by-case basis before applying a DAF of 20 to sources larger than 0.5 acre.

2.7 Mass-Limit Model Development

This section describes the development of models to solve the mass-balance violations inherent in the infinite source (i.e., steady-state) models used to calculate SSLs for the migration to ground water exposure pathway. The model developed is not a finite source model per se, but is designed for use with the current infinite source (i.e., steady-state) model to provide a lower, mass-based limit for SSLs for the migration to ground water.

The mass-limit model calculates a soil concentration that corresponds to the release of all radionuclides present within the source, at a constant health-based concentration, over the duration of exposure. These concentration limits are used as a minimum concentration for each SSL; below this concentration, a receptor point concentration time-averaged over the exposure period cannot exceed the health-based concentration on which it is based.

2.7.1 Mass Balance Issues. Infinite source (i.e., steady-state) models are subject to mass balance violations under certain conditions. Depending on a compound's solubility and the size of the source, modeled leaching rates can result in a source being depleted in a shorter time than the exposure duration (or the flux over a 30- or 70-year duration would release a greater mass of radionuclides than are present). Several commenters to the December 1994 draft Soil Screening Guidance for chemicals expressed concern that it is unrealistic for total emissions over the duration of exposure to exceed the total mass of contaminants in a source.

In summary, the mass-limit approach offers the following advantages:

- It corrects the possible mass-balance violation in the infinite-source SSLs.
- It does not require development of a finite source model to calculate SSLs.
- It is appropriate for screening, being based on the conservative assumption that all of the radionuclide present leaches over the period of exposure.
- It is easy to develop and implement, requiring only very simple algebraic equations and input parameters that are, with the exception of source depth, already used to calculate SSLs.

The derivation of this model is described below. It should be noted that the American Industrial Health Council (AIHC) independently developed identical models to solve the mass-balance violation as part of their public comments on the Soil Screening Guidance.

2.7.2 Migration to Ground Water Mass-Limit Model. For the migration to ground water pathway, the mass of radionuclide leached from a contaminant source over a fixed exposure duration (ED) period can be calculated as

$$M_l = C_w * I * A_s * ED \tag{39}$$

where

M_l = mass of contaminant leached (g)
C_w = leachate contaminant concentration (mg/L or g/m^3)
I = infiltration rate (m/yr)
A_s = source area (m^2)
ED = exposure duration (yr).

The total mass of contaminants present in a source can be expressed as

$$M_T = C_t * \rho_b * A_s * d_s \tag{40}$$

where

M_T = total mass of contaminant present (g)
C_t = total soil contaminant concentration (mg/kg or g/Mg, dry basis)
ρ_b = dry soil bulk density (kg/L or Mg/m^3)
A_s = source area (m^2)
d_s = source depth (m).

To avoid a mass balance violation, the mass of contaminant leached cannot exceed the total mass of contaminants present (i.e., M_l cannot exceed M_T). Therefore, the maximum possible contaminant mass that can be leached from a source (assuming no volatilization or degradation) is M_T and the upper limit for M_l is

$$M_l = M_T$$

or

$$C_w * I * A_s * ED = C_t * \rho_b * A_s * d_s \tag{41}$$

Rearranging to solve for the total soil concentration (C_t) corresponding to this situation (i.e., maximum possible leaching), as equation 41 becomes:

Mass-Limit Model for Migration to Ground Water Pathway

$$C_t = \frac{C_w * I * ED}{\rho_b * d_s} \tag{42}$$

Parameter/Definition (units)	Default
C_t/screening level in soil (mg/kg)	--
C_w/target soil leachate concentration (mg/L)	(MCL or HBL) × 20 DAF
I/infiltration rate (m/yr)	site-specific
ED/exposure duration (yr)	70
ρ_b/dry soil bulk density (kg/L)	1.5
d_s/average source depth (m)	site-specific

This soil concentration (C_t) represents a lower limit for soil screening levels calculated for the migration to ground water pathway. It represents the soil concentration corresponding to complete release of soil contaminants over the ED time period at a constant soil leachate concentration (C_w). Below this C_t, the soil leachate concentration averaged over the ED time period cannot exceed C_w.

SSLs for radionuclides may also be calculated in traditional radiological units of pCi/g, as follows:

$$SSL = \frac{C_w * I * ED * 1x10^{-3}}{\rho_b * d_s} \tag{43}$$

Parameter/Definition (units)	Default
SSL/screening level in soil (pCi/g)	--
C_w/target soil leachate concentration (pCi/L)	(MCL or HBL) × 20 DAF
I/infiltration rate (m/yr)	site-specific
ED/exposure duration (yr)	70
1x10-3/conversion factor (kg/g)	--
ρ_b/dry soil bulk density (kg/L)	1.5
d_s/average source depth (m)	site-specific

Part 3: UNSATURATED ZONE MODELS FOR RADIONUCLIDE FATE AND TRANSPORT

In an effort to provide useful information for model application, EPA's NRMRL/SPRD in Ada, Oklahoma, conducted an evaluation of five unsaturated zone fate and transport models for radionuclides. The results of this effort follow. The models reviewed are only a subset of the potentially appropriate models available to the public and are not meant to be construed as having received EPA approval. Other models also may be applicable to SSL development, depending on site-specific circumstances.

Each of the unsaturated zone models selected for evaluation are capable, to varying degrees, of simulating the transport and transformation of chemicals in the subsurface. Even the most unique site conditions can be simulated by either a single model or a combination of models. However, the intended uses and the required input parameters of these models vary. The models evaluated include:

- HYDRUS
- MULTIMED_DP 1.0
- FECTUZ
- CHAIN
- CHAIN 2D

The applications, assumptions, and input requirements for the five models evaluated are described in this section. The model descriptions include model solution method (i.e., analytical, numerical), purpose of model, and methods used by the model to simulate water flow, decay reaction, and radionuclide transport. Each description is accompanied by a table of required input parameters. Input parameters discussed include soil properties, chemical properties, meteorological data, and other site information. In addition, certain input control parameters may be required, such as time stepping, grid discritization information, and output format. Information on determining the general applicability of the models to subsurface conditions is provided, followed by an assessment of each model's potential applicability to the soil screening process.

HYDRUS. Information on the HYDRUS model (Version 6.0) was obtained from Šimůnek et al. (1998). HYDRUS is a finite-element model for one-dimensional solute and heat transport simulations in variably saturated media. The flow equation incorporates a sink term to account for water uptake by plant roots. The boundary conditions for flow, and heat and solute transport can vary with time. A finite source also can be modeled. Soil parameters are described by the van Genuchten parameters. The model also considers hysteresis in the water movement. Solute transport and transformation incorporates molecular diffusion, hydrodynamic dispersion, linear equilibrium reactions between the liquid and gaseous phases, nonlinear nonequilibrium partitioning (sorption) between the solid and liquid phases, zero-order production, and first-order decay/degradation reaction. The input parameters required by HYDRUS are presented in Table 3.1.

HYDRUS considers up to six solutes/radionuclides in the transport equation. For decay reaction, the radionuclides can be either coupled in a unidirectional chain or may move independently of each other. In the former case, two or more radionuclides are coupled in a sequential chain reaction. In the later case, each decay reaction is independent of each other. A specific distribution coefficient (or soil-water partition coefficient, a K_d value) as well as a specific decay rate for each radionuclide is required for the HYDRUS model simulation.

Table 3.1. Input Parameters Required for HYDRUS

Soil properties	Site characteristics	Pollutant properties	Root uptake parameters
Number of soil materials	Uniform or stepwise rainfall intensity	Molecular diffusion coefficient	Potential transpiration rate
Depth of soil layers	Volumetric fraction of solid phase	Dispersivity	Osmotic coefficient
Saturated water content	Volumetric fraction of organic matter	Freundlich isotherm coefficients‡	Pressure head where transpiration is reduced by 50%
Residual water content	Thermal dispersivity	Freundlich isotherm exponents‡	Root density as a function of depth
Saturated hydraulic conductivity	Empirical parameters for thermal conductivity calculations	First order rate constants (dissolved)	Power function in stress-response function (van Genuchten or Feddes)
Soil bulk density	Volumetric heat capacities of solid phase, organic matter, and liquid phase	First order rate constants (adsorbed)	--
van Genuchten retention parameter, α	Number of solutes	Decay coefficient (dissolved)	--
van Genuchten retention parameter, β	Contaminant concentrations in soil	Decay coefficient (adsorbed)	--
Rescaling factors for hydraulic properties	--	--	--
Heat transport properties			
Volumetric solid phase fraction	Coefficients of the thermal conductivity functions	Volumetric heat capacity of liquid phase for each soil	Longitudinal and transverse thermal dispersivities for each soil
Volumetric organic fraction of each soil	Volumetric heat capacity of solid phase for each soil	Volumetric heat capacity of organic matter for each soil	Thermal initial and boundary conditions

‡ Freundlich isotherm becomes a linear isotherm and Freundlich isotherm coefficient becomes a distribution coefficient (Kd value) when Freundlich isotherm exponent is 1.

MULTIMED_DP. Information on the MULTIMED_DP 1.0 (Multimedia exposure assessment model, including fate and transformation products) model was obtained from Liu et al. (1995), Salhotra et al. (1995) and Sharp-Hansen et al. (1995). MULTIMED_DP was initially developed as a multimedia fate and transport model to simulate contaminant migration from a waste disposal unit (MULTIMED) through different pathways in air, surface water, soil, and ground water. It has been modified to simulate the transport and fate of first and second-generation transformation products through the unsaturated and saturated zones. In MULTIMED_DP Version 1.0, the model now has an option to allow it to be used for unsaturated zone transport alone. The capacity for Monte Carlo simulation has been extended to the unsaturated zone, as well as to the saturated zone. Contamination of a surface stream due to the complete interception of a steady-state saturated zone plume is simulated by the surface water module. The air emissions and the atmosphere dispersion modules simulate the movement of chemicals into the atmosphere. For this review, only the fate and transport of pollutants from the soil to the ground water pathway is considered in detail. In MULTIMED_DP, infiltration of waste into the unsaturated or saturated zones can be simulated using a landfill module or by direct infiltration to the unsaturated or saturated zones. The unsaturated flow model simulates a one-dimensional steady flow with a semi-analytical solution and includes the option to consider seasonal variability in precipitation and evapotranspiration while retaining the assumption of steady-state. Transport in the unsaturated zone considers the effects of advection, dispersion, linear or nonlinear sorption, volatilization, hydrolysis, biodegradation, and first-order chemical decay. It can address steady or time variable infiltration and a finite or infinite source. A one-dimensional uniform steady flow in the saturated zone is assumed. The saturated transport module is also one-dimensional, but considers three-dimensional dispersion, linear adsorption, first-order decay, and dilution due to recharge. Mixing in the underlying saturated zone is based on a specified vertical dispersivity, the length of the disposal facility parallel to the flow direction, the thickness of the saturated zone, the ground water velocity, and the infiltration rate. The parameters required for the unsaturated and saturated zone transport in MULTIMED_DP are presented in Table 3.2.

The MULTIMED_DP model can handle parent, daughter, and granddaughter species chain decay. The chain decay reactions, for example, can be either one parent with one or two daughters or one parent with one daughter and two granddaughters. Effective/overall decay rate, individual decay rates , and distribution coefficient are required for the parent, daughter, and granddaughter species.

Table 3.2. Input Parameters Required for MULTIMED

Site characteristics/Source characteristics		
Recharge rate	Length scale of facility	Initial parent concentration at landfill
Infiltration rate	Width scale of facility	Initial daughter concentration at landfill
Depth of unsaturated zone	Duration of pulse	Initial granddaughter concentration at landfill
Area of waste disposal unit	Source decay constant	--
	Unsaturated zone parameters	
Number of physical flow layers	Residual water content	Distribution coefficient
Thickness of each layer	pH of layer	Temperature of layer
Number of porous materials	Soil bulk density	Brooks and Corey exponent
Saturated hydraulic conductivity	Longitudinal dispersivity	Air entry pressure head
van Genuchten retention parameter, α	Reference temperature for air diffusion	--
van Genuchten retention parameter, β	–	–

Table 3.2. Input Parameters Required for MULTIMED

	Saturated zone parameters	
Hydraulic gradient	Organic carbon content	Parent retardation coefficients
Hydraulic conductivity	Longitudinal dispersivity	Daughter retardation coefficients
Mixing zone depth	Transverse dispersivity	Granddaughter retardation coefficient
Aquifer thickness	Vertical dispersivity	Well distance from site
Aquifer porosity	Temperature of aquifer	Angle off-center of well
Bulk density	pH	Well Vertical distance

Table 3.2. Input Parameters Required for MULTIMED

Parent/Daughter/Granddaughter Decay Module

- Parent to daughter 1 dissolved Phase decay rate constant
- Parent to daughter 2 dissolved phase decay rate constant
- General parent dissolved phase decay rate constant (decay not contributing to either daughter product)
- Parent to daughter 1 sorbed phase decay rate constant
- Parent to daughter 2 sorbed phase decay rate constant
- General parent sorbed phase decay rate constant (decay not contributing to either daughter product)
- Daughter to granddaughter j, j+1 dissolved phase decay rate constant
- Daughter I general dissolved phase sorbed phase decay rate constant
- Daughter I to granddaughter j, j+1 sorbed phase decay rate constant
- Daughter I general sorbed phase decay rate constant
- Daughter second-order acid-catalysis hydrolysis rate constants
- Granddaughter k dissolved phase decay rate constant
- Granddaughter k sorbed phase decay rate constant
- Parent general term second-order acid-catalysis hydrolysis rate constant at reference temperature
- Parent general term neutral hydrolysis rate constant at reference temperature
- Parent general term second-order base-catalysis hydrolysis rate constant at reference temperature
- Parent to daughter 1 second-order acid catalysis hydrolysis rate constant at reference temperature
- Parent to daughter 1 neutral hydrolysis rate constant at reference temperature
- Parent to daughter 1 base-catalysis hydrolysis rate constant at Reference Temperature
- Parent general term neutral hydrolysis rate constant at reference temperature
- Parent general term second-order base-catalysis hydrolysis rate constant at reference temperature
- Parent general term second-order base-catalysis hydrolysis rate constant at reference temperature
- Daughter second-order acid-catalysis hydrolysis rate constants
- Daughter neutral catalysis hydrolysis rate constants
- Daughter second-order base-catalysis hydrolysis rate constant
- Granddaughter second-order acid catalysis hydrolysis rate constant
- Granddaughter neutral hydrolysis rate constant
- Granddaughter second-order base-catalysis hydrolysis rate constant
- Parent to daughter I dissolved and sorbed phase stoichiometric coefficients
- Daughter to granddaughter m dissolved and sorbed phase stoichiometric coefficients
- pH of the unsaturated zone or saturated zone
- Reference temperature
- Temperature of the unsaturated or saturated zone

FECTUZ. FECTUZ is a one-dimensional fate and transport model for the unsaturated zone which is incorporated into the EPACMTP program (EPA 1995b, 1995c). It is an extension of the VADOFT code (Huyakorn and Buckley, 1987) which is also incorporated into the RUSTIC and PRZM codes. FECTUZ simulates migration of contaminants from a landfill (or a surface impoundment), through the unsaturated zone to an unconfined aquifer with a water table present at some depth. The model allows for finite or infinite sources which can undergo decay using Bateman's equation (EPA, 1995b). The model can simulate linear and nonlinear adsorption and first order decay. FECTUZ is limited to the simulation of simple hydrogeological sites.

FECTUZ is capable of chain decay with up to seven radionuclide species (one parent with up to six daughter products). The chain decay path can be either straight or branched. A daughter product (a species) can have one or more immediate parent species. The sum of the chemical and biological transformation coefficients gives the overall decay coefficient. The fraction of each parent that decays into the same daughter product has to be specified. A distribution coefficient and a decay rate are specified for each individual radionuclide. A radionuclide species with a zero decay rate indicates it is the end product in the decay chain.

FECTUZ assumes steady state flow in the unsaturated zone which can be composed of one or more uniform soil layers. It simulates the transport of contaminants in soil using a advective-dispersive equation. FECTUZ employs three solution options for the transport equation. An analytical solution is used for steady state single species decay with linear adsorption. A semi-analytical solution is used for transient and steady state chain decay and linear sorption. A finite element solution is used for chain decay with nonlinear sorption. The input parameters required for FECTUZ are presented in Table 3.3.

Table 3.3. Input Parameters Required for FECTUZ Module within EPACMTP

Soil properties	Site characteristics	Pollutant properties
Soil bulk density	Thickness of unsaturated zone	Organic carbon partition coefficient
Saturated water content	Uniform thickness for discretized soil layers	Freundlich isotherm coefficients
Saturated hydraulic conductivity	Uniform infiltration rate except for surface impoundments	Dispersivity
Residual water content	Constant source or Decaying source or finite pulsed source	Decay coefficient (dissolved)
van Genuchten retention parameter, α	--	Decay coefficient (adsorbed)
van Genuchten retention parameter, β	--	Parent and daughter species decay reaction stoichiometry
Fraction of organic carbon	--	--

‡ Freundlich isotherm becomes a linear isotherm and Freundlich isotherm coefficient becomes a distribution coefficient (Kd value) when Freundlich isotherm exponent is 1.

CHAIN. CHAIN (van Genuchten, 1985) is a simple program which uses analytical solutions to solve for the simultaneous one-dimensional advective-dispersive transport of contaminants for up to four members of a sequential (first-order) decay chain. The model assumes that the soil system is a homogeneous soil system and that the moisture content and infiltration are constant in time (steady state flow). The adsorbed concentrations of the contaminant are represented by linear reversible isotherms. The transport contaminant boundary condition is either a constant concentration or a source decay given an arbitrary general release mechanism or the specific release mechanism defined by the Bateman equations (van Genuchten, 1985). The analytical solutions provide the solution in terms of concentration vs. distance (depth) at selected times as well as concentrations vs. time at selected depths. The input parameters required for the CHAIN model are presented in Table 3.4.

The consecutive chain reaction included in the transport model can involve up to four species. For example, a reaction could have parent and daughter products (two species) or a reaction could involve parent, daughter, and granddaughter products (three species). There is a decay rate associated with each pair of parent-daughter (or daughter-granddaughter) products. For the last product in the consecutive chain, a decay rate of zero is specified.

Table 3.4. Input Parameters Required for CHAIN Model

Soil properties	Site characteristics	Pollutant properties
Soil Bulk density	Source initial concentrations	Retardation values for decay chain members
Volumetric water content	Bateman coefficients for source decay	First order decay coefficients for decay chain members
Pore velocity (flux/ water content)	General release parameters for source decay	Pulse time
--	Dispersion coefficient	--

CHAIN 2D. CHAIN 2D is a two-dimensional model for the simulation of variably saturated flow, contaminant transport, and heat transport developed by Šimůnek and van Genuchten (1994**)**. The water flow is represented by Richards equation for saturated and unsaturated flow. It contains a sink term for water uptake by plant roots. The flow region can be composed of non-uniform soils to incorporate the effects of anisotropy. The flow model boundary conditions can include prescribed head, gradient, flux boundaries, or free drainage and a simplified representation of nodal drains. Heat and contaminant transport are modeled by the advective-dispersive equation which includes conduction and convection processes. The contaminant transport simulation can be modified for nonlinear nonequilibrium reactions between solid and liquid phases and for linear equilibrium reactions between liquid and gaseous phases. The contaminant transport simulation also includes zero order production and two first-order decay reactions: one which is independent of other solutes, and one which solutes are in sequential chain decay reactions. CHAIN 2D can simulate up to six species independent of one another or in a unidirectional chain decay. The boundary conditions for the contaminant transport could be constant concentration or constant flux. Flow and transport can be simulated in a vertical or horizontal plane or in an axisymmetrical cylindrical system. The flow and transport equations are solved using the Galerkin finite element method. The parameters required for CHAIN 2D model are presented in Table 3.5.

Similar to the HYDRUS model, CHAIN 2D considers up to six solutes/radionuclides in the transport equation, The radionuclide can be either coupled in a unidirectional chain or may move independently of each other. In the former case, two or more radionuclides are coupled in a sequential chain reaction. In the latter case, each decay reaction is independent of each other. A specific distribution coefficient as well as a specific decay rate for each radionuclide is required for the model simulation.

Table 3.5. Input Parameters Required for CHAIN 2D

Soil properties	Site characteristics	Pollutant properties	Root uptake parameters
2D cell discritization	Transpiration rate	Ionic or molecular diffusion coefficient in free water for each species	Root density as a function of depth
Saturated water content	Evaporation/infiltration rates	Ionic or molecular diffusion coefficient in gas phase for each species	Power function in stress-response function
Saturated hydraulic conductivity	Initial contaminant concentrations in soil	Longitudinal and transverse dispersivities for each species	Pressure head where transpiration is reduced by 50%
Soil bulk density	Contaminant species initial and boundary conditions	First order decay coefficient for each species in liquid, solid or gas phase	--
van Genuchten retention parameter, α	Initial head conditions	Zero order rate constant for each species in liquid, solid or gas phase	--
van Genuchten retention parameter, β	Location and rates of pumping/ injection wells	Adsorption (Freundlich)‡ isotherm coefficients for each species	--
Residual water content	Seepage faces, tile drains	Source Decay	--
Heat transport properties			
Volumetric solid phase fraction	Coefficients of the thermal conductivity functions	Volumetric heat capacity of liquid phase for each soil	Longitudinal and transverse thermal dispersivities for each soil
Volumetric organic fraction of each soil	Volumetric heat capacity of solid phase for each soil	Volumetric heat capacity of organic matter for each soil	Thermal initial and boundary conditions

‡ Freundlich isotherm becomes a linear isotherm and Freundlich isotherm coefficient becomes a distribution coefficient (Kd value) when Freundlich isotherm exponent is 1.

3.1 Considerations for Unsaturated Zone Model Selection. The accuracy of a model in a site-specific application depends on simplifications and assumptions implicit in the model and their relationship to site-specific conditions. Errors may be introduced from assumptions made when deriving input parameters. Although each of the five models evaluated has been tested and validated for simulation of water and radionuclide movement in the unsaturated zone, they are different in purpose and complexity, with certain models designed to simulate very specific scenarios. A model should be selected to accommodate a site-specific scenario as closely as possible. For example, if contaminant chain decay is of concern, the model should consider chain decay reaction. After a model is determined to be appropriate for a site, the contaminant(s) and the conditions to be modeled, the site-specific information available (or potentially available) should be compared to the input requirements for the model to ensure that adequate inputs can be developed. The unsaturated zone models addressed in this study use either analytical, semianalytical, or numerical solution methods. Analytical models represent the simplest models, requiring the least number of input parameters. They use a closed-form solution for the pertinent equations. In analytical models, certain assumptions have to be made with respect to the geometry of the system and external stresses. For this reason, there are few analytical flow models (van der Heijde, 1994). Analytical solutions are common for fate and transport problems which arise from the solutions of advective-dispersive equations. Analytical models (e.g. CHAIN) may require the assumption of uniform flow conditions, both spatially and temporally. Semianalytical models (e.g., MULTIMED_DP) approximate complex analytical solutions using numerical techniques (van der Heijde, 1994). Transient or steady-state conditions can be approximated using a semianalytical model. However, spatial variability in soil or aquifer conditions are usually not accommodated. Numerical models (e.g., HYDRUS or CHAIN-2D) use approximations of pertinent partial differential equations, such as finite-difference or finite-element methods. The resolution of the area and time of simulation is defined by the modeler. Numerical models may be used when simulating time-dependent scenarios, spatially variable soil conditions, and unsteady flow (van der Heijde, 1994).

3.2 Model Applicability to SSLs for Radionuclides. Evaluation of model applicability to SSLs is based on the following considerations:

1. Whether a model can be used in simulating fate and transport of the five selected radionuclides -- uranium, strontium, technetium, plutonium, and tritium?
2. Whether a model can simulate fate and transport of the selected radionuclides for a test case (the Las Cruces Trench Site, New Mexico)?
3. Is a model capable for use in the soil screening process?
4. What are the limitations for a specific model?

To facilitate the evaluation, the characteristics and capabilities of all five models are summarized in Table 3.6. These characteristics/capabilities include flow and transport processes, site-specific conditions, solution methods, assumptions, and model outputs. Each characteristic/capability is evaluated and is check marked when that characteristic/capacity is implemented in the model. Note that Table 3.6 addresses only unsaturated zone fate and transport model components in spite of the HYDRUS, MULTIMED_DP, and FECTUZ models having saturated zone flow and transport capabilities. The following text highlights a general description of the evaluation process, discusses some of the differences between the models, outlines their advantages and disadvantages, and describes appropriate scenarios for model application.

<u>Model Applicability to the Five Selected Radionuclides</u>: As provided in Table 3.6, all five models address four essential processes that predominately control the migration of radionuclides in the unsaturated zone -- advection (derived from infiltration), dispersion, sorption and radionuclide (straight chain and/or branched) decay. Therefore, **any of the five models can be used in simulating fate and transport for the five selected radionuclides --** uranium, strontium, technetium, plutonium, and tritium under the flow and site-specific conditions (e.g., steady-flow

in uniform soil, or transient flow in a layered soil) which have been implemented in the model. These five radionuclides exhibit a broad range of physical properties in soil -- distribution coefficients (K_d) of a radionuclide between water solution and soil, and chain decay rates. The simulation results for all five radionuclides using any of the five models would be similar to the results in the sensitivity analysis for K_d when the simulation time is much smaller than the magnitude of the half-life of any radionuclide.

Model Applicability to Site-Specific Conditions at the Las Cruces Trench Site: At the Las Cruces Trench Site, soil properties and solute transport parameters are available (Wierenga et al. 1991; Hills et al. 1991). They are reproduced in Tables 3.7 and 3.8. Other site-specific conditions are given in Table 3.9. As indicated in Table 3.6, the HYDRUS, CHAIN 2D, FECTUZ, and MULTIMED_DP models can simulate layered soils. However, **when soil at the site is treated as a uniform soil and the averaged soil properties are used, all five models can simulate the transport of radionuclides under steady-state water flow conditions at the site (assuming net recharge rate is known)**. Only HYDRUS and the CHAIN 2D model can handle transient flow at the site. Among the five models evaluated, CHAIN does not have a water flow module, FECTUZ considers water flow with a known infiltration rate, and the HYDRUS, CHAIN 2D and MULTIMED_DP models can calculate infiltration rates from rainfall and potential evaporation data. Note that the average annual infiltration rate at a site is difficult to measure in the field, yet it is required for estimating a dilution factor or DAF in the simplified SSL method. Furthermore, it is worthy to note that the processes/characteristics listed in Table 3.6 are not inclusive enough to accurately simulate the fate and transport of radionuclides in unsaturated zone. Other processes such as colloid-facilitated transport, pH-facilitated transport, cation-exchange, or complexation, are not considered. These processes might significantly influence the radionuclide movement in clay-rich soil, but they are not likely at the Las Cruces Trench Site, where the soils are predominantly classified as sands, sandy loams, loamy sands and sandy clay loams (Wierenga et al. 1991). For the purpose of SSL processes, all five models can simulate the transport of a radionuclide to a reasonable extent because these models consider the four essential processes that probably control the migration of radionulides in the unsaturated zone -- advection (derived from infiltration), dispersion, sorption, and radionuclide (straight and/or branched) decay. This will be further confirmed in the sensitivity analysis.

Model Capability for the Use in the Soil Screening Process: All five unsaturated models for radionuclides evaluated herein are PC-based, public domain models. These five models can calculate the leachate radionuclide concentrations entering ground water (Table 3.10). The leachate concentrations are needed in the SSL process/comparison. The leachate concentrations as a function of time are used to estimate ground water concentrations at the receptor well. These receptor point concentrations are then compared with the acceptable ground-water concentration (e.g. MCL) to determine if a site's soil exceed SSLs. If they do not exceed the acceptable ground-water concentrations, it might be believed that there is no concern for human health and ecology at the site. The conclusion of such a comparison is based on the assumptions that --1) the conceptual site model (CSM) is reasonably developed; 2) site-specific data required by the model are properly collected and meet data quality standards; and 3) the unsaturated zone model for radionuclide migration is properly used. Therefore, all five selected unsaturated zone models for radionuclides -- CHAIN, HYDRUS, CHAIN 2D, MULTIMED_DP, and FECTUZ, are basically **capable for use in the soil screening levels process**. When there is a concern for the uncertainty of the SSL estimates due to variability of the input parameters, the built-in Monte Carlo simulation in the MULTIMED_DP and FECTUZ (EPACMTP version) can be employed. For CHAIN, HYDRUS, and CHAIN 2D models, multiple runs using a set of variable input parameters (either obtained from the data survey or data generation for a know distribution of a parameter.) is needed.

Using any of these five models for the SSL calculation requires more site-specific data and modeling effort than using the simple site-specific SSL calculation (see Section 2 in TBD). However, these five models (and other potential models) can take into account more complex site conditions and can provide more accurate SSL calculation if data collection is sufficient and model application is properly done. In other words, using models in the SSL process require less assumptions than using the simple site-specific SSL calculation. However, for both the simple approach and the more detailed modeling approach the receptor well is assumed to be located at the edge of the waste source. The SSLs are intended to correspond to levels of radionuclides in soil such that MCLs will not be exceeded in the underlying ground water beyond the edge of the waste area. Among those simplifying assumptions listed in the Highlight 2 of TBD Section 2, only the assumptions of ignoring facilitated transport and complexation of nuclides with other constituents in soil solution still apply to the model SSL calculation.

Note that two methods can be used to calculate the receptor point concentration using the leachate radionuclide concentrations to ground water provided by the unsaturated model. One method uses a dilution/attenuation factor (DAF) to account for the mixing of leachate with ambient ground water. Another uses a saturated zone flow and transport model to calculate the resulting concentration considering the processes of advective, dispersive, decay, and biodegradation of the radionuclide in ground water (the MULTIMED_DP and FECTUZ models have a saturated zone flow and transport component.) When the former method is used, the receptor well is at the edge of the source (i.e., there is no dilution from recharge downgradient of the site) and is screened within the plume. A default value of 20 for DAF as proposed in the technical background document (TBD) for SSLs for radionuclide can be used. Alternatively, the dilution models in the section 2.6.4 in TBD can also be used.

It is worthy to note that in using the simple soil/water partition equation, the SSLs are directly calculated from an acceptable radionuclide level at the receptor well. The existing total soil radionuclide concentrations are then compared with the SSLs (TBD, Section 2.6). However, the unsaturated zone models do not perform this direct calculation. Instead, the existing site radionuclide-contamination is simulated, and the resulting radionuclide concentrations are then compared to the acceptable radionuclide level at the receptor well.

As described above, all five models can simulate the migration of radionulides in the unsaturated zone to some degree of satisfaction, depending on the complexity of the site-specific conditions. The final aspect for model applicability considered here is the complexity of the model and its ease-of-use. As given in Table 3.6, CHAIN is the simplest and easiest model while the rest are approximately the same in model complexity and ease-of-use. The pre- and post-processors for EPACMTP (FECTUZ) and MULTIMED (MULTIMED_DP) would provide some degree of help in the use of the model. Lastly, usability of the model user's manuals is also very important for model applicability, but it is not evaluated here. The following sections provide the description of the model applicability in the SSL processes. A summary table is also provided to highlight the key points in the model application to the SSL processes (Table 3.10).

HYDRUS. The HYDRUS model can simulate chemical movement in layered soils from a finite source. It may be useful in settings where low-permeability clay layers may attenuate contaminants through adsorption. The model also considers root zone water uptake and evapotranspiration so that infiltration (net recharge) into the soil can be obtained from rainfall data. The net infiltration amount needs to be provided by the user for the models CHAIN, FECTUZ, and MULTIMED-DP. HYDRUS outputs the radionuclide concentration in the soil water as a function of time and depth (including at the water table), which can be used in the SSLs comparison, along with the amount of chemical remaining in the soil. The model also outputs cumulative solute flux across the bottom of the soil profile (water table). Because it can estimate infiltration from rainfall intensities, HYDRUS may be useful in SSL applications. Grid discretization for HYDRUS version 6.0 requires extra effort. Availability of root water uptake parameters and potential evaporation might be limited at a site if the root water uptake process is considered.

MULTIMED-DP. MULTIMED-DP simulates simple vertical water movement in the unsaturated zone. Because an initial soil concentration cannot be specified, either the soil/water partition equation or a leaching test (SPLP) must be used to estimate soil leachate contaminant concentrations. MULTIMED_DP is appropriate for simulating contaminant migration in soil and can be used to model vadose zone attenuation of leachate concentrations derived from a partition equation. MULTIMED_DP outputs the leachate radionuclide concentration at the water table which is needed for the SSLs comparison. In addition, because it links the output from the unsaturated zone transport module with a saturated zone module, it can be used to determine the concentration of a radionuclide in a receptor well. MULTIMED_DP is appropriate for early-stage site simulations because the input parameters required are typically available (a database of chemical properties is provided in MULTIMED_DP). This model is the only model which considers runoff. Therefore, when runoff is an important hydrological process at a site, selection of MULTIMED_DP might be considered. Uncertainty analyses can be performed using Monte Carlo simulations for those parameters for which reliable values are not known. Because of the complexity of the model and requirement of a great amount of input data, expertise in properly using the model for SSLs is essential.

FECTUZ. FECTUZ at one time was a stand-alone unsaturated zone model, but, in recent years the model is only available as one of the coupled modules in the EPACMTP program (EPA 1995b, 1995c). FECTUZ simulates migration of contaminants from a landfill (or a surface impoundment), with finite or infinite sources, through the unsaturated zone and into ground water. The model allows time varying precipitation data for inputs, but a steady-state water flow is assumed during each precipitation event. The concentrations of radionuclides at a specific time and space are part of the model outputs and can be used in the SSL process. The implementation of Monte Carlo simulation in the EPACMTP program also provides the capacity of uncertainty analysis for the FECTUZ model. It is worthy to note that mixed units for the model inputs (e.g. cm/hr, m/year, cm, m) are used in FECTUZ. Therefore, caution should be taken for the use of correct values and units in FECTUZ.

CHAIN. The CHAIN model is simple and easy to use in simulating fate and transport of radionuclides in a uniform soil under steady unsaturated flow conditions. The model outputs the leachate radionuclide concentrations at the specified time and depth (including at the water table) and these concentrations can be used in the SSL process. It is not adequate to use CHAIN for fate and transport of a radionuclide in layered soil. However, the CHAIN model might be used in a layered soil by using representative uniform soil properties as an approximation of heterogeneous soil properties. The CHAIN model could be the first choice as a preliminary assessment tool in SSLs evaluating for radionuclides.

CHAIN 2D. Basically, CHAIN 2D is very similar to HYDRUS 6.0. The major difference is it is a two-dimensional model. When the assumption of vertical flow in the unsaturated zone is in question, CHAIN 2D may be the choice to simulate leaching of a radionuclide from the disposal facility to ground water. This is especially true when the leaching area is small and horizontal flow becomes significant in a highly stratified soil. The CHAIN 2D model outputs the radionuclide concentration in the soil water as a function of time and depth, which can be used in the SSLs comparison, along with the amount of chemical remaining in the soil. The model also outputs cumulative solute flux across the bottom of the soil profile (water table). Because it can estimate infiltration from rainfall, the model may be useful in SSL applications. Both the CHAIN 2D and HYDRUS models are capable of simulating water uptake by plant roots. Both models can be used for estimating net recharge when potential evaporation and water uptake parameters are available.

Examples of Model Application in the Soil Screening Processes

To demonstrate how to apply the five models for radionuclides evaluated herein in the SSL estimation, a conceptual site model was developed at the Las Cruces Trench Site and the base case simulations were performed. It is assumed that the Site had been used as a waste disposal/storage facility where radionuclides from tank leaks or improper waste disposal were released to the soil surface for 1000 days with a total amount for $3x10^{-4}$ mg /cm^2 ^{99}Tc (^{99}Tc concentration from the waste source is 1.25×10^{-2} mg/L)**.** Rainfall infiltration (with a net annual recharge rate of 87 mm/y) is the driving force for the downward migration of radionuclide to the water table beneath. Base values of the input parameter are given in Table 3.11. It is assumed that the steady-state uniform water flow occurs at the site.

Time-varied leachate concentrations of ^{99}Tc entering ground water predicted by the CHAIN, HYDRUS, MULTIMED_DP, FECTUZ, and CHAIN 2D models are presented in Figures 3.1 and 3.2. The results indicate that the five models provide similar breakthrough curves except that numerical dispersion is observed using MULTIMED_DP. Figures 1 and 2 also show how variability of the distribution coefficient (K_d) influence the migration of ^{99}Tc. Increasing Kd values would reduce migration of radionuclides. The breakthrough curves for the decay (daughter) product ^{99}Ru, using the CHAIN and FECTUZ (EPACMTP), are presented in Figure 3.3. The breakthrough curves predicted by the CHAIN and FECTUZ models are not distinguishable. Using MULTIMED_DP, a large numerical dispersion was observed (not shown). It was also found that the breakthrough curves predicted by the CHAIN model will exhibit a great oscillation if single precision is used in the CHAIN code. Figure 3.3 is also used to demonstrate the decay reaction implemented in the five models. No attempt was made in using the HYDRUS and CHAIN 2D models for obtaining the breakthrough curves of the decay products since simulation times of up to 800 years are required.

For the use of models in the SSL process, the task is to simulate the conceptual site scenario and examine whether the leachate concentration entering the ground water will result in an exceedance of the radionuclide concentration over the MCL at the receptor well. The results shown in Figure 3.4 indicate that the calculated concentrations at the receptor well exceed the MCL (5.3E-5 mg/L) for ^{99}Tc. In other words, the site conditions exhibit a soil radionuclide contamination of ^{99}Tc exceeding the SSL and thus has imposed a potential risk to human health and the environment. Note a few things about Figure 3.4: 1) three receptor well locations (0 m, 10 m and 30 m from the waste source) were examined; 2) two methods were employed to calculate the dilution effect of the radionuclides in the saturated zone once they enter the water table. One was the use of the simple dilution factor. Another method was the use of a saturated zone model (SZM) such as FECTUZ/EPACMTP to simulate the radionuclide transport. Using the dilution factor, a default DAF value of 20 as well as the dilution factors, at three ground-water velocities (5 m/y, 10 m/y, and 25 m/y), obtained from equations 7 and 8 in the section 2.5.2 of the SSG user's guide were used. The parameters used for the dilution factor calculation and the ground water flow and transport simulation using FECTUZ/EPACMTP are given in Table 3.12.

The details of the model application in the SSL process and the sensitivity analysis are provided in the report entitled "*Evaluation of Computer Models for Simulating Radionuclide Transport in the Unsaturated Zone*"

Table 3.6. Summary Comparisons of the Vadose Zone Model for Radionuclides in the SSL Process

Model component	HYDRUS	MULTIMED-DP	FECTUZ	CHAIN	CHAIN 2D
Contaminants					
Organics	!	!	!	!	!
Metals	!	!	!	!	!
Radionuclides (parent)	!	!	!	!	!
Radionuclides (progeny)	!	!	!	!	!
Non-aqueous phase liquids	--	--	--	--	--
Site characteristics					
Vadose zone					
Layered soil	!	!	!	--	!
Sources types					
Contaminated soil	!	!	!	!	!
Landfill	!	!	!	--	!
Surface impoundment	--	--	!	--	--
Waste piles	--	--	!	--	--
Source term characteristics					
Mass balance	!	!	!	!	!
Multimedia partitioning	!	!	!	--	!
Source decay	--	!	!	!	--
Source ingrowth (radionuclides)	--	--	--	--	--
Multiple contaminants per simulations	!	!	!	!	!
Source release mechanisms					
Leaching	!	!	!	!	!
Direct release to:					
Vadose zone	!	!	!	!	!
Groundwater	--	!	!	--	--
Surface water	--	!	--	--	--
Air	--	!	--	--	--
Medium-specific flow					
Air Box model (0-D, complete mixing)	--	--	--	--	--
Surface Hydrology					
Precipitation	--	!	--	--	--
Runoff	--	!	--	--	--
Infiltration	!	!	!	!	!
ET	!	!	--	--	!
Surface Water (Stream discharge)	--	!	--	--	--
Vadose Zone					
Vadose zone (Steady-state infiltration -->soil	!	!	!	!	!
Vadose zone (n-D dynamic)	!	--	--	--	!
Groundwater	--	!	!	--	--
Medium-specific contaminant transport					
Atmosphere (emission through diffusion)	--	!	--	--	--
Surface water (stream interception and mixing)	--	!	--	--	--
Vadose zone (1-D advection and dispersion)	!	!	!	!	!
Vadose zone (2-D advection and dispersion)	--	--	--	--	!
Groundwater					
Homogeneous aquifer (1-D advection and dispersion)	--	!	!	--	--
Homogeneous aquifer (2-D advection and dispersion)	--	!	!	--	--
Homogeneous aquifer (3-D advection and dispersion)	--	!	!	--	--
Medium-specific heat transport	!	--	--	--	!

Contaminant transformations and fate processes					
1st order decay (not decay products)	!	!	!	!	!
1st order decay (with chained daughter and granddaughter decay products) --straight chain	!	!	!	!	!
1st order decay -- branch chain	!	--	!	--	!
Non-1st order decay	!	--	--	--	!
Linear partitioning (water/soil)	!	!	!	!	!
Nonlinear partitioning (water/soil)	!	--	!	--	!
Hydrolysis	--	!	!	--	--
Chemical reactions/speciation	–	!	!	--	--
Intermedia contaminant fluxes					
Surface soil --> Air (volatilization)	!	!	--	--	!
Surface soil --> Vadose zone (leaching)	!	!	!	!	!
Surface soil --> Overland (erosion, runoff)	--	!	--	--	--
Surface water -->Sediment (sedimentation)	--	--	--	--	--
Vadose zone --> groundwater (percolation)	!	!	!	!	!
Vadose zone --> Air (volatilization)	!	!	--	--	!
Groundwater --> Surface water (deposition)	--	--	--	--	--
Air --> Surface soil (deposition)	--	--	--	--	--
Air --> Surface water (deposition)	--	--	--	--	--
Exposure pathways	--	--	--	--	--
Human health endpoints	--	--	--	--	--

! denotes component is included in model; -- denotes component is not included in model.

Table 3.7. Soil Hydraulic Properties at the Las Cruces Trench Site for SSG Model Evaluation Study (Modified from Wierenga et al., 1991)

Layers	Depth (cm)	Saturated water content (cm^3/cm^3)	Residual water content (cm^3/cm^3)	van Genuchten alpha coefficient, α, (cm^{-1})	van Genuchten beta coefficient, β (--)	Saturated hydraulic conductivity, K_s, (cm/d)
Uniform Soil Model						
all	0-600	0.321	0.083	0.055	1.509	270
Layered Soil Model						
1	0-15	0.348	0.095	0.042	1.903	539
2	15-140	0.343	0.091	0.062	1.528	250
3	140-205	0.336	0.085	0.060	1.574	267
4	205-250	0.313	0.071	0.068	1.537	300
5	250-305	0.302	0.072	0.040	1.550	250
6	305-370	0.294	0.090	0.070	1.711	334
7	370-460	0.310	0.073	0.027	1.418	221
8	460-540	0.325	0.083	0.041	1.383	172
9	540-600	0.306	0.078	0.047	1.432	226

Table 3.8. Solute Transport Properties at the Las Cruces Trench Site for SSG Model Evaluation Study (Modified from Porro and Wierenga, 1993)

Layers	Depth (cm)	Pore velocity, v, (cm/d)	Dispersion coefficient, D, (cm^2/d)	Dispersivity, ε, (cm)
Uniform Soil Model				
all	0-500	14.7	62.1	4.53
Layered Soil Model				
1	82	18.4	40.5	2.20
2	125	16.5	66.1	4.00
3	220	17.1	52.5	3.06
4	310	13.2	67.3	5.09
5	400	10.9	84.6	7.80
6	500	12.3	61.9	5.04

Table 3.9. Characteristics of the Las Cruces Trench Site for SSG Model Evaluation Study (modified from Gee et al., 1994)

Annual Precipitation (cm/y)	Annual Potential (Pan) Evaporation (cm/y)	Annual Potential Recharge (cm/y)	Average Daily Max. Air Temperature (^{O}C)	Average Daily Min. Air Temperature (^{O}C)	Elevation (m)	Depth to Water Table (m)	Geology	Typical Soil Type	Typical Vegetation
33.8	239	8.7	28	13	1357	60	Alluvial	Berino fine loamy sand	Creosote bush

Table 3.10. Summary of the Use of the Unsaturated Zone Models for Radionuclides in the SSL Process

Model	Processes, outputs, components
HYDRUS	- provides the leachate radionuclide concentrations entering the ground water so whether the resulting concentration of the radionuclide at the receptor well would exceed the acceptable level or not, can be examined - calculates infiltration which can be used as inputs in the SSL calculation - considers soil heterogeneity, time-varying infiltration and evapotranspiration - outputs radionuclide concentration in soil, cumulative flux across water table - grid discretization for HYDRUS version 6.0 requires extra effort
MULTIMED_DP	- provides the leachate radionuclide concentrations entering the ground water so whether the resulting concentration of the radionuclide at the receptor well would exceed the acceptable level or not, can be examined - uncertainty of model outputs can be examined - considers runoff, evapotranspiration - linked with a saturated flow and transport model - requires a great amount of input data, expertise because of model complexity
FECTUZ	- provides the leachate radionuclide concentrations entering the ground water so whether the resulting concentration of the radionuclide at the receptor well would exceed the acceptable level or not, can be examined - uncertainty of model outputs can be examined - linked with a saturated flow and transport model - uses mixed units for the input data
CHAIN	- provides the leachate radionuclide concentrations entering the ground water so whether the resulting concentration of the radionuclide at the receptor well would exceed the acceptable level or not, can be examined - used for simplified radionuclide-contaminated site scenario - simple-to-use, less input data requirement - as a preliminary assessment tool in SSL estimation
CHAIN 2D	- provides the leachate radionuclide concentrations entering the ground water so whether the resulting concentration of the radionuclide at the receptor well would exceed the acceptable level or not, can be examined - calculates infiltration which can be used as inputs in the SSL calculation - considers soil heterogeneity, time-varying infiltration and evapotranspiration - outputs radionuclide concentration in soil, cumulative flux across water table - considers two-dimensional soil heterogeneity

Table 3.11. Base Values of Input Parameters for the Unsaturated Zone Models

Parameters	Values
Source-Specific Parameters	
Area of disposal facility (m^2)	400
Width of disposal facility (m)	20
Length of disposal facility (m)	20
Mass release of (parent) radionuclide ^{99}TC (mg/cm^2)	$3x10^{-4}$
Concentration of ^{99}Tc in recharge water from waste source (mg/L)	$1.25x10^{-2}$
Duration of waste source being completely released (days)	1000
Net recharge rate (mm/y)	87
Water content (cm^3/cm^3)	0.16
Source decay constant	--
Soil Properties in Unsaturated Zone (Uniform Soil in Table 3-7)	
Saturated hydraulic conductivity, K_s, (cm/d)	270.1
Porosity (--)	0.358
Residual water content (cm^3/cm^3)	0.083
Saturated water content (cm^3/cm^3)	0.321
Bulk density	1.70
van Genuchten alpha coefficient, α, (cm^{-1})	0.055
van Genuchten beta coefficient, β (--)	1.509
Depth to water table (m)	6
Solute Transport Parameters	
Decay coefficient for parent (^{99}TC) (1/y)	$3.3x10^{-6}$
Decay coefficient for daughter (^{99}Ru) (1/y)	$7.9x10^{-9}$
Distribution coefficient for parent (^{99}TC) (ml/g)	0.007
Distribution coefficient for daughter (^{99}Ru) (ml/g)	5.0
Dispersion Coefficient (cm^2/d)	1.0
Dispersivity (cm)	4.53

Table 3.12. Input Parameters Used for the Dilution Factor Calculation and Transport Simulation of Radonuclide in the Saturated Zone using FECTUZ/EPACMTP Model

Parameters	Values
Hydraulic Properties in Saturated Zone	
Aquifer hydraulic conductivity (m/y)	500
Hydraulic gradient (m/m)	0.005
Porosity	0.32
Bulk density (g/cm^3)	1.70
Longitudinal dispersivity (m)	1.0
Transverse dispersivitiy (m)	0.2
Vertical dispersivity (m)	0.04
Aquifer Characteristics	
Aquifer (m)	10
Computed Dilution Factor Using Equations 7 & 8 in the Section 2.5.2 of the SSG User's Guide	
Dilution factor for ground-water velocity = 5 m/y	8.1
Dilution factor for ground-water velocity = 10 m/y	14.2
Dilution factor for ground-water velocity = 25 m/y	32.4

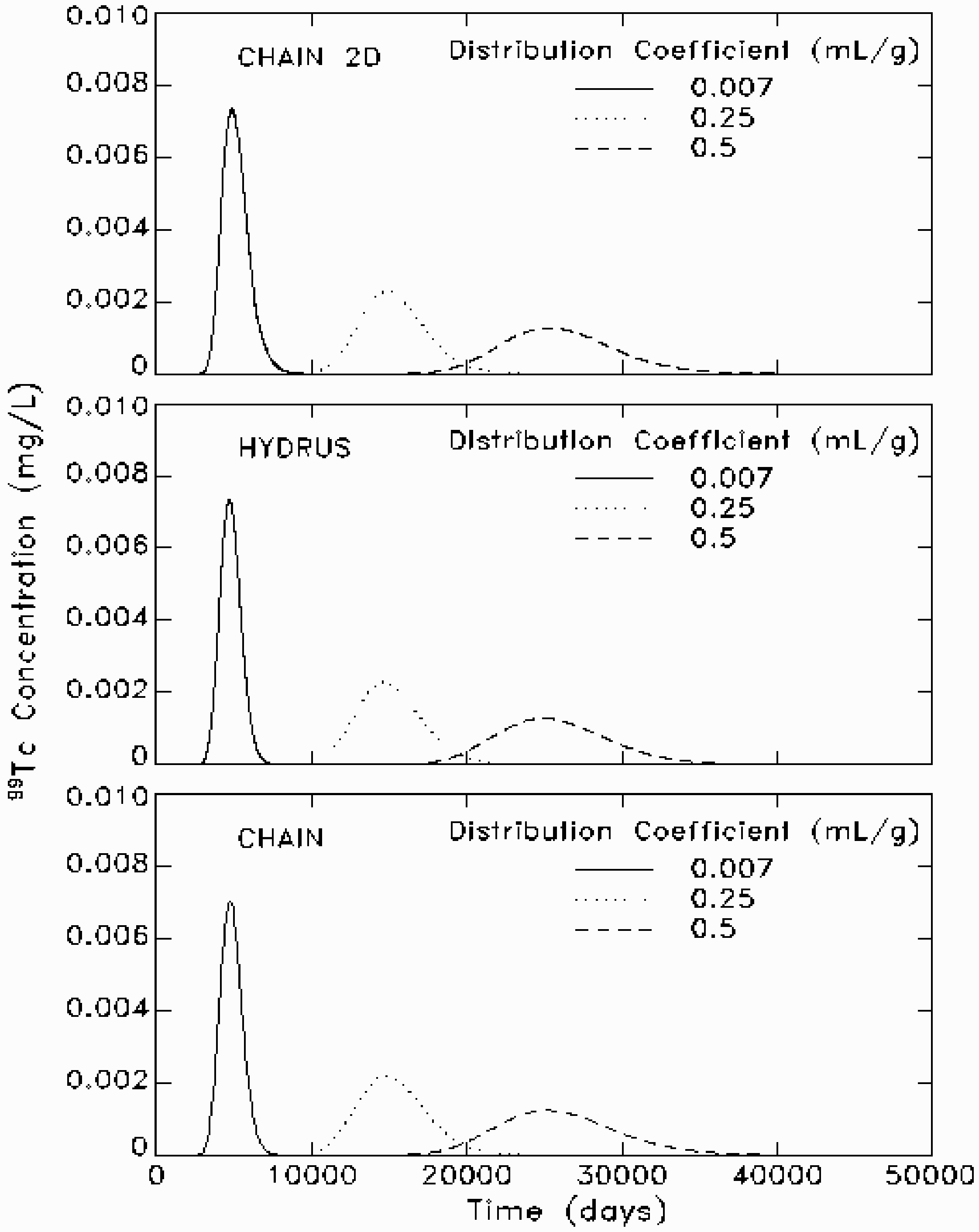


Figure 3.1. Base case simulations of ^{99}TC breakthrough (through the unsaturated zone) with three distribution coefficients using the CHAIN, HYDRUS, and CHAIN 2D models.

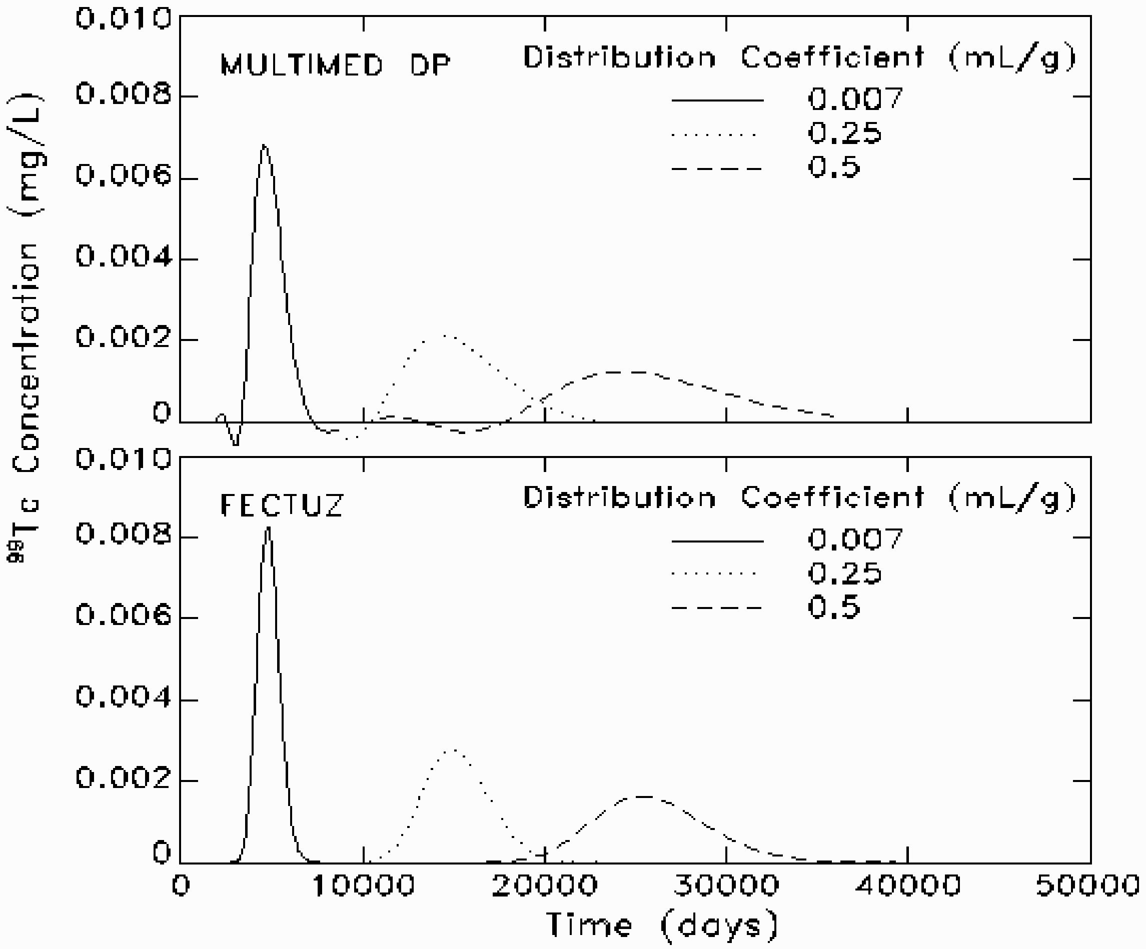


Figure 3.2. Base case simulations of ^{99}TC breakthrough (through the unsaturated zone) with three different distribution coefficients using the FECTUZ, and MULTIMED_DP models.

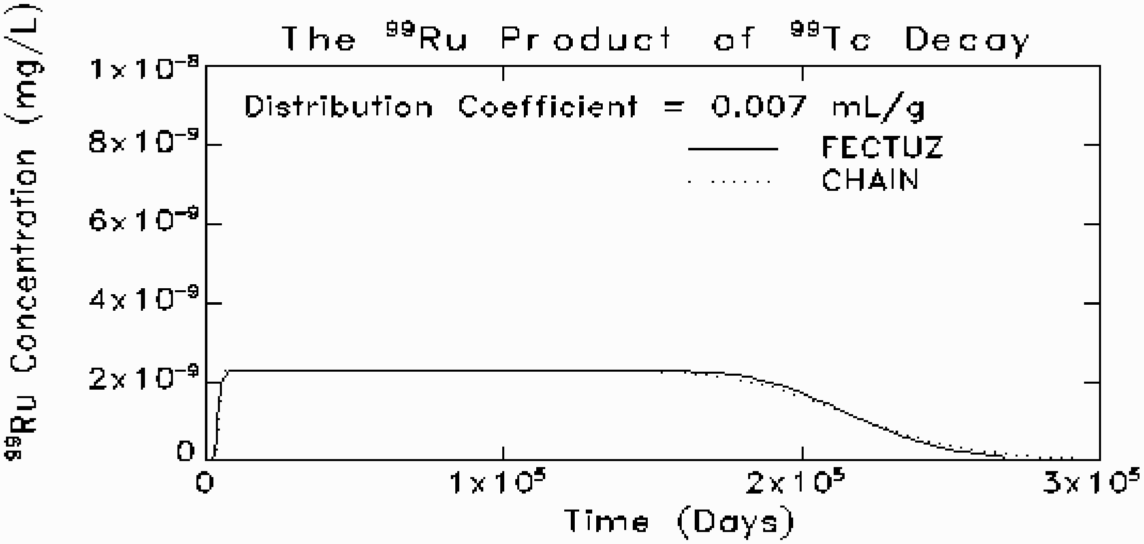


Figure 3.3. Base case simulations of the daughter product -- ^{99}Ru of ^{99}TC breakthrough (through the unsaturated zone) with a distribution coefficient of 0.007 mL/g using FECTUZ, and CHAIN models.

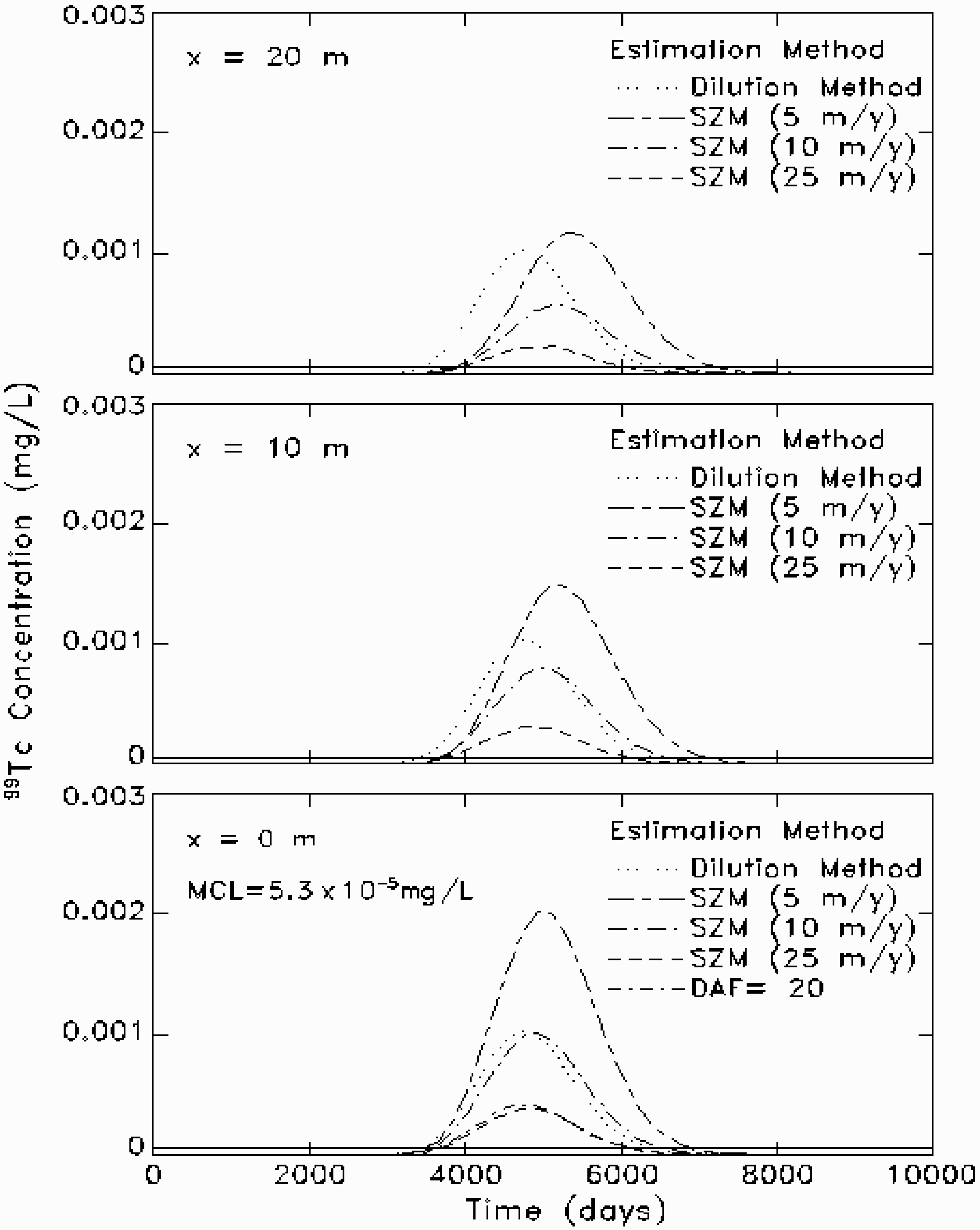


Figure 3.4. Base case predication of ^{99}TC concentration at the receptor well by employing dilution factor method and using saturated zone model (SZM) -- FECTUZ. The numbers in the parenthes2s are ground water velocity. The solid lines are used to represent the MCL (5.3 x 10^{-5} ml/L) level.

Part 4: MEASURING RADIONUCLIDE CONCENTRATIONS IN SOIL

The Soil Screening Guidance for Radionuclides includes a sampling strategy for implementing the soil screening process. Section 4.1 presents the sampling approach for surface soils. This approach provides a simple decision rule based on comparing the maximum radionuclide concentrations of composite samples with surface soil screening levels (the Max test) to determine whether further investigation is needed for a particular exposure area (EA). In addition, this section presents a more complex strategy (the Sign test) that allows the user to design a site-specific quantitative sampling strategy by varying decision error limits and soil radionuclide variability to optimize the number of samples and composites. Section 4.2 provides a subsurface soil sampling strategy for developing SSLs and applying the screening procedure for the migration to ground water exposure pathways.

Section 4.3 describes the technical details behind the development of the SSL sampling strategy, including analyses and response to public and peer-review comments received on the December 1994 draft Soil Screening Guidance for chemicals.

The sampling strategy for the soil screening process is designed to achieve the following objectives:

- Estimate mean concentrations of radionuclides of concern for comparison with SSLs
- Fill in the data gaps in the conceptual site model necessary to develop SSLs.

The soils of interest for the first objective differ according to the exposure pathway being addressed. For the direct ingestion of soil, inhalation of fugitive dust, external radiation exposure, and ingestion of homegrown produce pathways, EPA is concerned about surface soils. The sampling goal is to determine average radionuclide concentrations of surface soils in exposure areas of concern. For the migration to ground water pathway, subsurface soils are the primary concern. For this pathway, the average radionuclide concentration through each source is the parameter of interest.

The second objective (filling in the data gaps) applies primarily to the external radiation exposure, inhalation of fugitive dust, ingestion of homegrown produce, and migration to ground water pathways. For these pathways, the source area and depth as well as average soil properties within the source are needed to calculate the pathway-specific SSLs. Therefore, the sampling strategy needs to address collection of these site-specific data.

Because of the difference in objectives, the sampling strategies for surface soils and subsurface soils are addressed separately. If both surface and subsurface soils are a concern, then surface soils should be sampled first because the results of surface soil analyses may help delineate source areas to target for subsurface sampling.

At some sites, a third sampling objective may be appropriate. As discussed in the Soil Screening Guidance, applying SSLs may be difficult at sites where background radionuclide levels are above the SSLs. Where sampling information suggests that background radionuclide concentrations may be a concern, background sampling may be necessary. *Methods for Evaluating the Attainment of Cleanup Standards - Volume 3: Reference-Based Standards for Soil and Solid Media* (U.S. EPA, 1994e) provides further information on sampling soils to determine background conditions at a site. The Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM) (U.S. EPA, 1997c) also provides specific guidance concerning site surveys when background radiation is a consideration.

In order to accurately represent radionuclide distributions at a site, EPA used the Data Quality Objectives (DQO) process (Figure 4) to develop a sampling strategy that will satisfy Superfund program objectives. The DQO process is a systematic data collection planning process developed by EPA to ensure that the right type, quality, and quantity of data are collected to support EPA decision making. As shown in Sections 4.1.1 through 4.1.6, most of the key outputs of the DQO process already have been developed as part of the Soil Screening Guidance. The DQO activities addressed in this section are described in detail in the *Data Quality Objectives for Superfund: Interim Final Guidance* (U.S. EPA, 1993b) and the *Guidance for the Data Quality Objectives Process* (U.S. EPA, 1994c). Refer to these documents for more information on how to complete each DQO activity or how to develop other, site-specific sampling strategies.

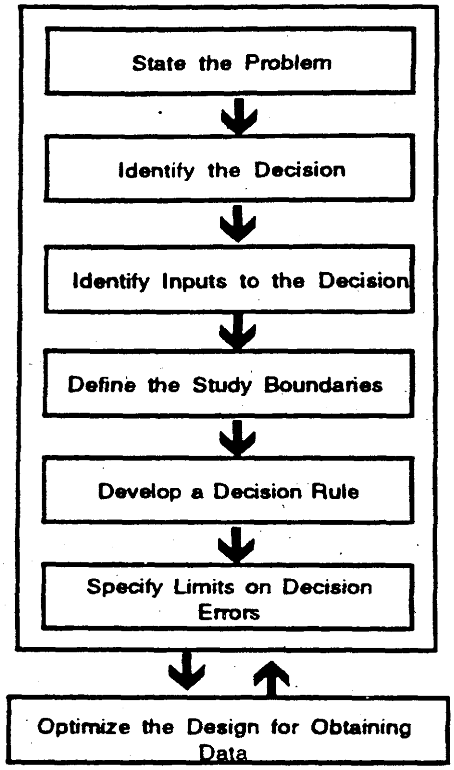


Figure 4. The Data Quality Objectives process.

4.1 Sampling Surface Soils

A sampling strategy for surface soils is presented in this section, organized by the steps of the DQO process. The first five steps of this process, from defining the problem through developing the basic decision rule, are summarized in Table 4.1, and are described in detail in the first five subsections. The details of the two remaining steps of the DQO process, specifying limits on decision errors and optimizing the design, have been developed separately for two alternative hypothesis testing procedures (the Max test and the Sign test) and are presented in four (4.1.6, 4.1.7, 4.1.9, and 4.1.10) subsections. In addition, a data quality assessment (DQA) follows the DQO process step for optimizing the design. The DQA ensures that site-specific error limits are achieved. Sections 4.1.8 and 4.1.11 describe the DQA for the Max and Sign tests, respectively. The technical details behind the development of the surface soil sampling design strategy are explained in Section 4.3.

4.1.1 State the Problem. In screening, the problem is to identify the radionuclides and exposure areas (EAs) that do not pose significant risk to human health so that future investigations can be focused on the areas and radionuclides of concern at a site.

The main site-specific activities involved in this first step of the DQO process include identifying the data collection planning team (including technical experts and key stakeholders) and specifying the available resources. The list of technical experts and stakeholders should contain all key personnel who are involved with applying the Soil Screening Guidance at the site. Other activities in this step include developing the conceptual site model (CSM), identifying exposure scenarios, and preparing a summary description of the surface soil contamination problem. The User's Guide for Radionuclides (U.S. EPA, 1999) describes these activities in with more detail.

4.1.2 Identify the Decision. The decision is to determine whether the mean surface soil concentrations exceed surface soil screening levels for specific radionuclides within EAs. If so, the EA must be investigated further. If not, no further action is necessary under CERCLA for the specific radionuclides in the surface soils of those EAs.

Table 4.1. Sampling Soil Screening DQOs for Surface Soils

DQO Process Steps	Soil Screening Inputs/Outputs
State the Problem	
Identify scoping team	Site manager and technical experts (e.g., health physicists, risk assessors, statisticians, soil scientists)
Develop conceptual site model (CSM)	CSM development (described in Step 1 of the User's Guide, U.S. EPA, 1996)
Define exposure scenarios	Direct ingestion of soil, inhalation of fugitive dusts, external radiation exposure, and ingestion of homegrown produce in a residential setting;
Specify available resources	Sampling and analysis budget, scheduling constraints, and available personnel
Write brief summary of contamination problem	Summary of the surface soil contamination problem to be investigated at the site
Identify the Decision	
Identify decision	Do mean soil concentrations for particular radionuclides (e.g., radionuclides of potential concern) exceed appropriate screening levels?
Identify alternative actions	Eliminate area from further study under CERCLA or Plan and conduct further investigation
Identify Inputs to the Decision	
Identify inputs	SSLs for each pathway for specified radionuclides Measurements of surface soil radionuclide concentration
Define basis for screening	Soil Screening Guidance for Radionuclides
Identify analytical methods	Feasible analytical methods (both field and laboratory) consistent with program-level requirements
Define the Study Boundaries	
Define geographic areas of field investigation	The entire NPL site (which may include areas beyond facility boundaries), except for any areas with clear evidence that no contamination has occurred
Define population of interest	Surface soils (usually the top 15 centimeters, but may be deeper where activities could redistribute subsurface soils to the surface)
Divide site into strata	Strata may be defined so that radionuclide concentrations are likely to be relatively homogeneous within each stratum based on the CSM and field measurements
Define temporal boundaries of study	Temporal constraints on scheduling field visits
Define scale of decision making	Exposure areas (EAs) no larger than 0.5 acre each (based on residential land use)
Identify practical constraints	Potential impediments to sample collection, such as access, health, and safety issues

Table 4.1. Sampling Soil Screening DQOs for Surface Soils

DQO Process Steps	Soil Screening Inputs/Outputs
Develop a Decision Rule	
Specify parameter of interest	?True mean" (μ) individual radionuclide concentration in each EA. (since the determination of the ?true mean" would require the collection and analysis of many samples, the ?Max Test" uses another sample statistic, the maximum composite concentration).
Specify screening level	Screening levels calculated using available parameters and site data (or generic SSLs if site data are unavailable).
Specify "if..., then..." decision rule	If the ?true mean" EA concentration exceeds the screening level, then investigate the EA further. If the ?true mean" is less than the screening level, then no further investigation of the EA is required under CERCLA.

4.1.3 Identify Inputs to the Decision. This step of the DQO process requires identifying the inputs to the decision process, including the basis for further investigation and the applicable analytical methods. The inputs for deciding whether to investigate further are the SSLs calculated for the site radionuclides for each pathway as described in Part 2 of this document, and the surface soil concentration measurements for those same radionuclides. Therefore, the remaining task is to identify radioanalytical laboratory methods and/or field methods for which the minimum detectable concentrations (MDCs) are less than the SSLs. EPA recommends the use of field methods where applicable and appropriate as long as quantitation limits are below the SSLs. At least 10 percent of field samples should be split and sent to a radioanalytical laboratory for confirmatory analysis (U.S. EPA, 1993d). However, for soil screening purposes, most SSLs for radionuclides are too low to be detected using typical radiological field methods. Therefore, for most radionuclides, sampling and laboratory analysis must be the primary means of soil screening for the majority of radionuclides.

4.1.4 Define the Study Boundaries. This step of the DQO process defines the sample population of interest, subdivides the site into appropriate exposure areas, and specifies temporal or practical constraints on the data collection. The description of the population of interest must include the surface soil depth.

Sampling Depth. When measuring soil chemical contamination levels at the surface for the ingestion and inhalation pathways, the Soil Screening Guidance for chemicals suggests that the top 2 centimeters is usually considered surface soil, as defined by *Urban Soil Lead Abatement Project* (U.S. EPA 1993f). It further recommends that additional sampling beyond this depth may be appropriate for surface soils under a future residential use scenario in areas where major soil disturbances can reasonably be expected as a result of landscaping, gardening, or construction activities. In this situation, contaminants that were at depth can be moved to the surface. Thus, it is important to be cognizant of local residential construction practices when determining the depth of surface soil sampling and to weigh the likelihood of that area being developed.

For the Soil Screening Guidance for Radionuclides, a surface soil depth of 15 cm has been assumed in order to ensure that SSLs calculated for radionuclides are adequately protective in a residential setting. The residential setting used to develop the SSLs for each radionuclide and pathway assumes that: 1) there is no clean cover of soil; 2) the top few centimeters of soil are available for resuspension in air; 3) the top 15 cm of contaminated soil are homogenized by agricultural activities (e.g., plowing); 4) there is a sufficiently large area and depth of contamination to approximate an infinite slab source for external exposure purposes; 5) there is enough land for a garden to produce homegrown fruits and vegetables for the residents; and 6) while the plant root system grows to a depth of 1 meter, most plant nutrients are obtained from within the upper 15 cm of soil.

Subdividing the Site. This step involves dividing the site into areas or strata depending on the likelihood of contamination and identifying areas with similar radionuclide patterns. These divisions can be based on process knowledge, operational units, historical records, and/or prior sampling. Partitioning the site into such areas and strata can lead to a more efficient sampling design for the entire site.

For example, the site manager may have documentation that large areas of the site are unlikely to have been used for waste disposal activities. These areas would be expected to exhibit relatively low variability and the sampling design could involve a relatively small number of samples. The greatest intensity of sampling effort would be expected to focus on areas of the site where there is greater uncertainty or greater variability associated with contamination patterns. When relatively large variability in radionuclide concentrations is expected, more samples are required to determine with confidence whether the EA should be screened out or investigated further.

Initially, the site may be partitioned into three types of areas:

1. Areas that are not likely to be contaminated
2. Areas that are known to be highly contaminated
3. Areas that are suspected to be contaminated and cannot be ruled out.

Areas that are not likely to be contaminated generally will not require further investigation if this assumption is based on historical site use information or other site data that are reasonably complete and accurate. (However, the site manager may also want take a few samples to confirm this assumption). These may be parts of the site that are within the legal boundaries of the property but were completely undisturbed by hazardous-waste-generating activities. All other areas need investigation.

Areas that are known to be highly contaminated (i.e., sources) are targeted for subsurface sampling. The information collected on source area and depth is used to calculate site-specific SSLs for the inhalation and migration to ground water pathways (see Section 4.2 for more information).

Areas that are suspected to be contaminated (and cannot be ruled out for screening) are the primary subjects of the surface soil investigation. If a geostatistician is available, a geostatistical model may be used to characterize these areas (e.g., kriging model). However, guidance for this type of design is beyond the scope of the current guidance (see Chapter 10 of U.S. EPA, 1989a).

Defining Exposure Areas. After the site has been partitioned into relatively homogeneous areas, each region that is targeted for surface soil sampling is then subdivided into EAs. An **exposure area** (EA) is a physical area of a specified size and shape for which a separate decision will be made as to whether or not the area exceeds the screening criteria. To facilitate sampling design and ensure that the number of sample data points for a specific site are relatively uniformly distributed among areas of similar contamination potential, the site is divided into EAs that share a common history or other characteristics, or are naturally distinguishable from other portions of the site. Because the SSLs were developed for a residential scenario, EPA assumes the EA is a suburban residential lot corresponding to 0.5 acre (2,000 m^2). For soil screening purposes, each EA should be 0.5 acre or less. To the extent possible, EAs should be constructed as square or rectangular areas that can be subdivided into squares to facilitate compositing and grid sampling. If the site is currently residential, then the EA should be the actual residential lot size. The exposure areas should not be laid out in such a way that they unnecessarily combine areas of high and low levels of contamination. The orientation and exact location of the EA, relative to the distribution of the radionuclide in the soil, can lead to instances where sampling of the EA may lead to results above the mean, and other instances, to results below the mean. Try to avoid straddling radionuclide ?distribution units" within the 0.5 acre EA.

The sampling strategy for surface soils allows investigators to determine mean soil radionuclide concentration across an EA of interest. An arithmetic mean concentration for an EA best represents the exposure to site radionuclides over a long period of time. For risk assessment purposes, an individual is assumed to move randomly across an EA over time, spending equivalent amounts of time in each location. Since reliable information about specific patterns of nonrandom activity for future use scenarios is not available, random exposure appears to be the most reasonable assumption for a residential exposure scenario. Therefore, spatially averaged surface soil concentrations are used to estimate mean exposure concentrations.

Because all the EAs within a given stratum should exhibit similar radionuclide concentrations, one site-specific sampling design can be developed for all EAs within that stratum. As discussed above, some strata may have relatively low variability and other strata may have relatively high variability. Consequently, a different sampling design may be necessary for each stratum, based upon the stratum-specific estimate of the radionuclide variability.

In addition to determining the mean concentration of a radionuclide in an EA, it is important to identify the presence of small areas of elevated activity. This is done by the performance of scanning surveys. The sensitivity of scanning surveys will be insufficient to detect small areas of elevated activity for most radionuclides with levels of contamination as low as those of the SSLs calculated for large areas of uniform contamination. However, standard scanning survey techniques may be able to detect SSLs calculated for smaller areas of contamination. Scan surveys are intended to provide a degree of confidence that any significant areas of elevated activity are identified. Therefore, scanning surveys should be performed for all EAs prior to sampling. The extent of the survey coverage should be dictated by the potential for small areas of elevated activity in the EA. EAs with a high potential for small areas of elevated activity should receive 100% coverage. In EAs with a very low potential for small areas of elevated activity, scanning surveys should be performed in at least 10% of the area. In such cases, the areas selected for scan should be those with highest potential based on professional judgement. Due to the limited sensitivity of scan surveys, any small areas of elevated activity found during the survey should be identified for further investigation (i.e., not screened out).

4.1.5 Develop a Decision Rule.

Ideally, the decision rule for surface soils is:

> If the mean radionuclide concentration within an EA exceeds the screening level, then investigate that EA further.

This "screening level" is the actual numerical value used to compare against the site contamination data. It may be identical to the SSL, or it may be a multiple of the SSL (e.g., 2 SSL) for a hypothesis test designed to achieve specified decision error rates in a specified region above and below the SSL. In addition, another sample statistic (e.g., the maximum concentration) may be used as an estimate of the mean for comparison with the "screening level."

4.1.6 Specify Limits on Decision Errors for the Max Test.

Sampling data will be used to support a decision about whether an EA requires further investigation. Because of variability in radionuclide concentrations within an EA, practical constraints on sample sizes, and sampling or measurement error, the data collected may be inaccurate or nonrepresentative and may mislead the decision maker into making an incorrect decision. A decision error occurs when sampling data mislead the decision maker into choosing a course of action that is different from or less desirable than the course of action that would have been chosen with perfect information (i.e., with no constraints on sample size and no measurement error).

EPA recognizes that data obtained from sampling and analysis are never perfectly representative and accurate, and that the costs of trying to achieve near-perfect results can outweigh the benefits. Consequently, EPA acknowledges that uncertainty in data must be tolerated to some degree. The DQO process controls the degree to which uncertainty in data affects the outcomes of decisions that are based on those data. This step of the DQO process allows the decision maker to set limits on the probabilities of making an incorrect decision.

The DQO process utilizes hypothesis tests to control decision errors. When performing a hypothesis test, a presumed or baseline condition, referred to as the "null hypothesis" (H_o), is established. This baseline condition is presumed to be true unless the data conclusively demonstrate otherwise, which is called "rejecting the null hypothesis" in favor of an alternative hypothesis. For the Soil Screening Guidance for Radionuclides, the baseline condition, or H_o, is that **the site needs further investigation.**

When the hypothesis test is performed, two possible decision errors may occur:

1. Decide not to investigate an EA further (i.e., "walk away") when the correct decision (with complete and perfect information) would be to "investigate further"

2. Decide to investigate further when the correct decision would be to "walk away."

Since the site is on the NPL, site areas are presumed to need further investigation. Therefore, the data must provide clear evidence that it would be acceptable to "walk away." This presumption provides the basis for classifying the two types of decision errors. The "incorrectly walk away" decision error is designated as the Type I decision error because one has incorrectly rejected the baseline condition (null hypothesis). Correspondingly, the "unnecessarily investigate further" decision error is designated as the Type II decision error.

To complete the specification of limits on decision errors, Type I and Type II decision error probability limits must be defined in relation to the SSL. First a "gray region" is specified with respect to the mean radionuclide concentration within an EA. The gray region represents the range of radionuclide levels near the SSL, where uncertainty in the data (i.e., the variability) can make the decision "too close to call." In other words, when the average of the data values is very close to the SSL, it would be too expensive to generate a data set of sufficient size and precision to resolve what the correct determination should be. (i.e., Does the average concentration fall "above" or "below" the SSL?)

The Soil Screening Guidance for Radionuclides establishes a default range for the width and location of the "gray region": from one-half the SSL (0.5 SSL) to two times the SSL (2 SSL). By specifying the upper edge of the gray region as twice the SSL, it is possible that exposure areas with mean values slightly higher than the SSL may be screened from further study. However, EPA believes that the exposure scenario and assumptions used to derive SSLs are sufficiently conservative to be protective in such cases.

On the lower side of the gray region, the consequences of decision errors at one-half the SSL are primarily financial. If the lower edge of the gray region were to be moved closer to the SSL, then more exposure areas that were truly below the SSL would be screened out, but more money would be spent on sampling to make this determination. If the lower edge of the gray region were to be moved closer to zero, then less money could be spent on sampling, but fewer EAs that were truly below the SSLs would be screened out, leading to unnecessary investigation of EAs. The Superfund program chose the gray region to be one-half to two times the SSL after investigating several different ranges. This range for the gray region represents a balance between the costs of collecting and analyzing soil samples and making incorrect decisions. While it is desirable to estimate exactly the exposure area mean, the number of samples required are much more than project managers are generally willing to collect in a "screening" effort. Although some exposure areas will have radionuclide concentrations that are between the SSL and twice the SSL and will be screened out, human health will still be protected given the conservative assumptions used to derive the SSLs.

The Soil Screening Guidance for Radionuclides establishes the following goals for Type I and Type II decision error rates:

- Prob ("walk away" when the true EA mean is 2 SSL) = 0.05
- Prob ("investigate further" when the true EA mean is 0.5 SSL) = 0.20.

This means that there should be no more than a 5 percent chance that the site manager will "walk away" from an EA where the true mean concentration is 2 SSL or more. In addition, there should be no more than a 20 percent chance that the site manager will unnecessarily investigate an EA when the mean is 0.5 SSL or less.

These decision error limits are general goals for the soil screening process. Consistent with the DQO process, these goals may be adjusted on a site-specific basis by considering the available resources (i.e., time and budget), the importance of screening surface soil relative to other potential exposure pathways, consequences of potential decision errors, and consistency with other relevant EPA guidance and programs.

Table 4.2 summarizes this step of the DQO process for the Max test, specifying limits on the decision error rates, and the final step of the DQO process for the Max test, optimizing the design. Figure 5 illustrates the gray region for the decision error goals: a Type I decision error rate of 0.05 (5 percent) at 2 SSL and a Type II decision error rate of 0.20 (20 percent) at 0.5 SSL.

4.1.7 Optimize the Design for the Max Test. This section provides instructions for developing an optimum sampling strategy for screening surface soils. It discusses compositing, the selection of sampling points for composited and uncomposited surface soil sampling, and the recommended procedures for determining the sample sizes necessary to achieve specified limits on decision errors using the Max test.

Note that the size, shape, and orientation of sampling volume (i.e., ?support") for heterogenous media have a significant effect on reported measurement values. For instance, particle size has a varying affect on the transport and fate of radionuclides in the environment and on the potential receptors. Because comparison of data from methods that are based on different supports can be difficult, defining the sampling support is important in the early stages of site characterization. This may be accomplished through the DQO process with existing knowledge of the site, contamination, and identification of the exposure pathways that need to be characterized. Refer to *Preparation of Soil Sampling Protocols: Sampling Techniques and Strategies* (U.S. EPA, 1992f) for more information about soil sampling support.

The SAP developed for surface soils should specify sampling and analytical procedures as well as the development of QA/QC procedures. To identify the appropriate analytical procedures, the screening levels must be known. If data are not available to calculate site-specific SSLs, then the generic SSLs in Appendix A should be used.

Compositing. Because the objective of surface soil screening is to ensure that the mean radionuclide concentration does not exceed the screening level, the physical "averaging" that occurs during compositing is consistent with the intended use of the data. Compositing allows a larger number of locations to be sampled while controlling analytical costs because several discrete samples are physically mixed (homogenized) and one or more subsamples are drawn from the mixture and submitted for analysis. If the individual samples in each composite are taken across the EA, each composite represents an estimate of the EA mean.

Table 4.2. Sampling Soil Screening DQOs for Surface Soils under the Max Test

DQO Process Steps	Soil Screening Inputs/Outputs
Specify Limits on Decision Errors*	
Define baseline condition (null hypothesis)	The EA needs further investigation
Define the gray region**	From 0.5 SSL to 2 SSL
Define Type I and Type II decision errors	Type I error: Do not investigate further ("walk away from") an EA whose true mean exceeds the screening level of 2 SSL Type II error: Investigate further when an EA's true mean falls below the screening level of 0.5 SSL
Identify consequences	Type I error: potential public health consequences Type II error: unnecessary expenditure of resources to investigate further
Assign acceptable probabilities of Type I and Type II decision errors	Goals: Type I: 0.05 (5%) probability of not investigating further when ?true mean” of the EA is 2 SSL Type II: 0.20 (20%) probability of investigating further when ?true mean” of the EA is 0.5 SSL
Define QA/QC goals	Radioanalytical laboratory precision and bias requirements 10% laboratory analyses for field methods
Optimize the Design	
Determine how to best estimate ?true mean”	Samples composited across the EA estimate the EA mean ($\bar{x}$). Use maximum composite concentration as a conservative estimate of the true EA mean.
Determine expected variability of EA surface soil radionuclide concentrations	A conservatively large expected coefficient of variation (CV) from prior data for the site, field measurements, or data from other comparable sites and expert judgment. A minimum default CV of 2.5 should be used when information is insufficient to estimate the CV.
Design sampling strategy by evaluating costs and performance of alternatives	Lowest cost sampling design option (i.e., compositing scheme and number of composites) that will achieve acceptable decision error rates
Develop planning documents for the field investigation	Sampling and Analysis Plan (SAP) Quality Assurance Project Plan (QAPjP)

* Since the DQO process controls the degree to which uncertainty in data affects the outcome of decisions that are based on that data, specifying limits on decision errors will allow the decision maker to control the probability of making an incorrect decision when using the DQOs.

** The gray region represents the area where the consequences of decision errors are minor (and uncertainty in sampling data makes decisions too close to call).

A practical constraint to compositing in some situations is the heterogeneity of the soil matrix. The efficiency and effectiveness of the mixing process may be hindered when soil particle sizes vary widely or when the soil matrix contains foreign objects, organic matter, viscous fluids, or sticky material. Soil samples should not be composited if matrix interference among radionuclides is likely (e.g., when the presence of one radionuclide biases analytical results for another).

Before individual specimens are composited for chemical analysis, the site manager should consider homogenizing and splitting each specimen. By compositing one portion of each specimen with the other specimens and storing one portion for potential future analysis, the spatial integrity of each specimen is maintained. If the concentration of a radionuclide in a composite sample is high, the splits of the individual specimens from which it was composed can be analyzed discretely to determine which individual specimen(s) have high concentrations of the radionuclide. This will permit the site manager to determine which portion within an EA is contaminated without making a repeat visit to the site.

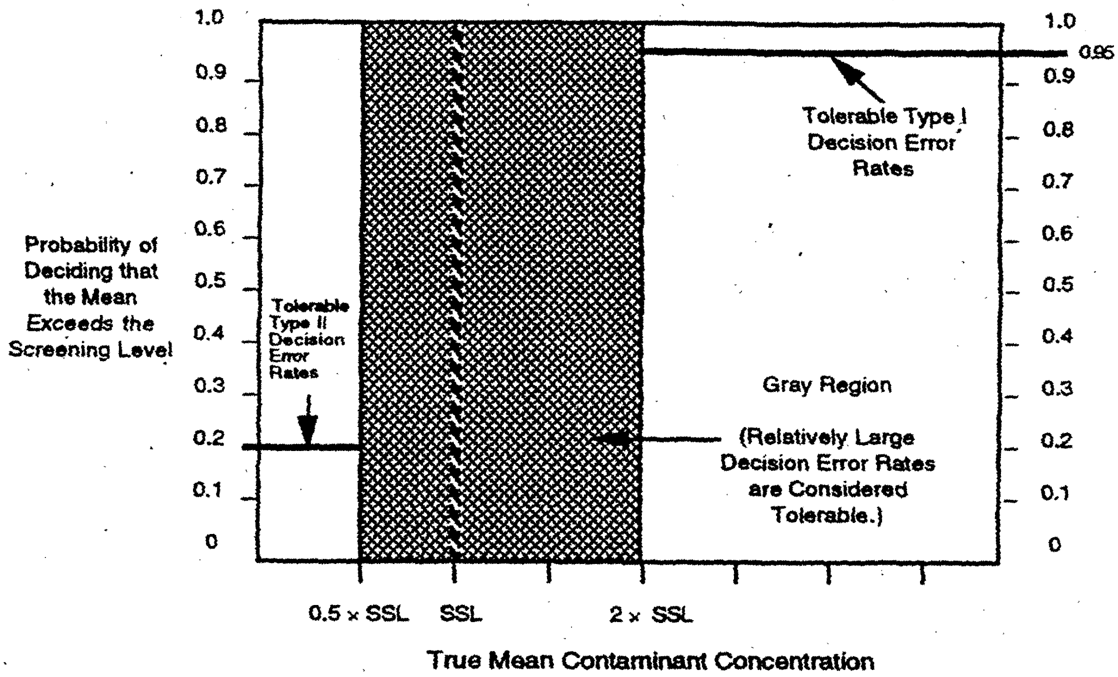


Figure 5. Design performance goal diagram.

Sample Pattern. The Max test should only be applied using composite samples that are representative of the entire EA. However, the Sign test (see Section 4.1.9) can be applied with individual, uncomposited samples. There are several options for developing a sampling pattern for compositing that produce samples that should be representative. If individual, uncomposited samples will be analyzed for radionuclide concentrations, the N sample points can be selected using either (1) simple random sampling (SRS), (2) stratified SRS, or (3) systematic grid sampling (square or rectangular grid) with a random starting point (SyGS/rs). Step-by-step procedures for selecting SRS and SyGS/rs samples are provided in Chapter 5 of the U.S. EPA (1989a) and Chapter 5 of U.S. EPA (1994e). If stratified random sampling is used, the sampling rate must be the same in every sector, or stratum of the EA. Hence, the number of sampling points assigned to a stratum must be directly proportional to the surface area of the stratum.

Systematic grid sampling with a random starting point is generally preferred because it ensures that the sample points will be dispersed across the entire EA. However, if the boundaries of the EA are irregular (e.g., around the perimeter of the site or the boundaries of a stratum within which the EAs were defined), the number of grid sample points that fall within the EA depends on the random starting point selected. Therefore, for these irregularly shaped EAs, SRS or stratified SRS is recommended. Moreover, if a systematic trend of contamination is suspected across the EA (e.g., a strip of higher contamination), then SRS or stratified SRS is recommended again. In this case, grid sampling would be likely to result in either over- or under representation of the strip of higher radionuclide levels, depending on the random starting point.

For composite sampling, the sampling pattern used to locate the discrete sample specimens that form each composite sample (N) is important. The composite samples should be formed in a manner that is consistent with the assumptions underlying the sample size calculations. In particular, each composite sample should provide an unbiased estimate of the mean radionuclide concentration over the entire EA. One way to construct a valid composite of C specimens is to divide the EA into C sectors, or strata, of equal area and select one point at random from each sector. If sectors (strata) are of unequal sizes, the simple average is no longer representative of the EA as a whole.

Five valid sampling patterns and compositing schemes for selecting N composite samples that each consist of C specimens are listed below:

1. Select an SRS consisting of C points and composite all specimens associated with these points into a sample. Repeat this process N times, discarding any points that were used in a previous sample.

2. Select an SyGS/rs of C points and composite all specimens associated with the points in this sample. Repeat this process N times, using a new randomly selected starting point each time.

3. Select a single SyGS/rs of C x N points and use the systematic compositing scheme that is described in Highlight 3 to form N composites, as illustrated in Figure 6.

4. Select a single SyGS/rs of C x N points and use the random compositing scheme that is described in Highlight 4 to form N composites, as illustrated in Figure 7.

5. Select a stratified random sample of C x N points and use a random compositing scheme, as described in Highlight 5, to form N composites, as illustrated in Figure 8.

Methods 1, 2, and 5 are the most statistically defensible, with method 5 used as the default method in the Soil Screening Guidance for Radionuclides. However, given the practical limits of implementing these methods, either method 3 or 4 is generally recommended for EAs with regular boundaries (e.g., square or rectangular). As noted above, if the boundaries of the EA are irregular, SyGS/rs sampling may not result in exactly C x N sample points. Therefore, for EAs with irregular boundaries, method 5 is recommended. Alternatively, a combination of methods 4 and 5 can be used for EAs that can be partitioned into C sectors of equal area of which K have regular boundaries and the remaining C - K have irregular boundaries.

Additionally, compositing within sectors to indicate whether one sector of the EA exceeds SSLs is an option that may also be considered. See Section 4.3.6 for a full discussion.

Sample Size. This section presents procedures to determine sample size requirements for the Max test that achieve the site-specific decision error limits discussed in Section 4.1.6. The Max test is based on the maximum concentration observed in N composite samples that each consist of C individual specimens. The individual specimens are selected so that each of the N composite samples is representative of the site as a whole, as discussed above. Hence, this section addresses determining the sample size pair, C and N, that achieves the site-specific decision error limits. Directions for performing the Max test in a manner that is consistent with DQOs established for a site are presented later in this section.

Table 4.3 presents the probabilities of Type I errors at 2 SSL and Type II errors at 0.5 SSL (the boundary points of the gray region discussed in Section 4.1.6) for several sample size options when the variability for concentrations of individual measurements across the EA ranges from 100 percent to 400 percent (CV = 1.0 to 4.0). Two choices for the number, C, of specimens per composite are shown in this table: 4 and 6. Fewer than four specimens per composite is not considered sufficient for the Max test. Fewer than four specimens per composite does not achieve the decision error limit goals for the level of variability generally encountered at CERCLA sites. More than six specimens may be more than can be effectively homogenized into a composite sample.

The number, N, of composite samples shown in Table 4.3 ranges from 4 to 9. Fewer than four samples is not considered sufficient because, considering decision error rates from simulation results (Section 4.3), the Max text should be based on at least four independent estimates of the EA mean. More than nine composite samples per EA is generally unlikely for screening surface soils at Superfund sites. However, additional sample size options can be determined from the simulation results reported in Appendix I.

Highlight 3: Procedure for Compositing of Specimens from a Grid Sample Using a Systematic Scheme (Figure 6)

1. Lay out a square or triangular grid sample over the EA, using a random start. Step-by-step procedures can be found in Chapter 4 of U.S. EPA (1989a). The number of points in the grid should be equal to C x N, where C is the desired number of specimens per composite and N is the desired number of composites.

2. Divide the EA into C sectors (strata) of equal area and shape such that each sector contains the same number of sample points. The number of sectors (C) should be equal to the number of specimens in each composite (since one specimen per area will be used in each composite) and the number of points within each sector, N, should equal the desired number of composite samples.

3. Label the points within one sector in any arbitrary fashion from 1 to N. Use the same scheme for each of the other sectors.

4. Form composite number 1 by compositing specimens with the '1' label, form composite number 2 by compositing specimens with the '2' label, etc. This leads to N composite samples that are subjected to chemical analysis.

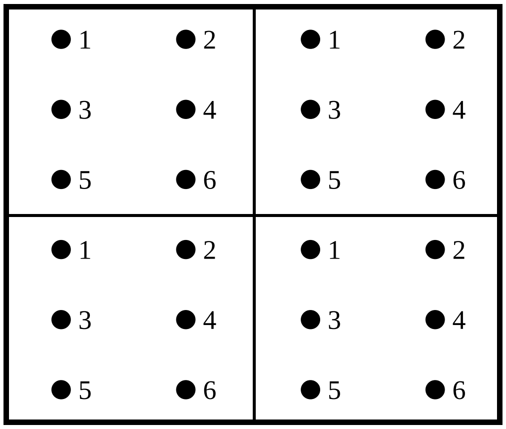


Figure 6. Systematic (square grid points) sample with systematic compositing scheme (6 composite samples consisting of 4 specimens).

Table 4.3. Probability of Decision Error at 0.5 SSL and 2 SSL Using Max Test

Sample Size[b]	CV = 1.0[a]		CV =1.5		CV = 2.0		CV = 2.5		CV = 3.0		CV = 3.5		CV = 4.0	
	$E_{0.5}$[c]	$E_{2.0}$[d]	$E_{0.5}$	$E_{2.0}$	$E_{0.5}$	$E_{2.0}$	$E_{0.5}$	$E_{2.0}$	$E_{0.5}$	$E_{2.0}$	$E_{0.5}$	$E_{2.0}$	$E_{0.5}$	$E_{2.0}$
	C = 4 specimens per composite[e]													
4	< .01	0.08	0.02	0.11	0.09	0.13	0.14	0.19	0.19	0.20	0.24	0.26	0.25	0.30
5	< .01	0.05	0.02	0.06	0.11	0.10	0.15	0.10	0.26	0.17	0.26	0.18	0.31	0.25
6	< .01	0.03	0.02	0.04	0.11	0.06	0.21	0.08	0.28	0.11	0.31	0.11	0.35	0.16
7	< .01	0.01	0.03	0.02	0.12	0.04	0.25	0.05	0.31	0.08	0.36	0.09	0.41	0.15
8	< .01	0.01	0.03	0.01	0.16	0.02	0.25	0.04	0.36	0.05	0.42	0.07	0.41	0.09
9	< .01	0.01	0.05	0.01	0.16	0.01	0.28	0.03	0.36	0.04	0.44	0.07	0.48	0.08
	C = 6 specimens per composite													
4	< .01	0.08	< .01	0.11	0.03	0.12	0.08	0.16	0.15	0.17	0.26	0.20	0.23	0.27
5	< .01	0.05	< .01	0.08	0.04	0.09	0.11	0.08	0.17	0.13	0.22	0.15	0.25	0.20
6	< .01	0.03	0.01	0.04	0.06	0.04	0.14	0.06	0.19	0.09	0.25	0.09	0.29	0.12
7	< .01	0.01	0.01	0.02	0.06	0.02	0.14	0.04	0.23	0.06	0.29	0.08	0.37	0.08
8	< .01	0.01	0.01	0.01	0.06	0.02	0.15	0.02	0.25	0.03	0.30	0.04	0.40	0.08
9	< .01	0.01	0.01	0.01	0.06	0.01	0.18	0.02	0.28	0.03	0.34	0.03	0.39	0.04

[a] The CV is the coefficient of variation for individual, uncomposited measurements across the entire EA, including measurement error

[b] Sample size (N) = number of composite samples

[c] $E_{0.5}$ = Probability of requiring further investigation when the EA mean is 0.5 SSL

[d] $E_{2.0}$ = Probability of requiring further investigation when the EA mean is 2.0 SSL

[e] C = number of specimens per composite sample, where each composite consists of points from a stratified random or systematic grid sample from across the entire EA

NOTE: All decision error rates are based on 1,000 simulations that assume each composite is representative of the entire EA, that half the EA has concentrations below the quantitation limit (i.e., SSL/100), and half the EA has concentrations that follow a gamma distribution (a conservative distributional assumption)

Highlight 4: Procedure for Compositing of Specimens from a Grid Sample Using a Random Scheme (Figure 7)

1. Lay out a square or triangular grid sample over the EA, using a random start. Step-by-step procedures can be found in Chapter 5 of U.S. EPA (1989a). The number of points in the grid should be equal to C x N, where C is the desired number of specimens per composite and N is the desired number of composites.

2. Divide the EA into C sectors (strata) of equal area and shape such that each sector contains the same number of sample points. The number of sectors (C) should be equal to the number of specimens in each composite (since one specimen per area will be used in each composite) and the number of points within each sector, N, should equal the desired number of composite samples.

3. Use a random number table or random number generator to establish a set of labels for the N points within each sector. This is done by first labeling the points in a sector in an arbitrary fashion (say, points A, B, C,...) and associating the first random number with point A, the second with point B, etc. Then rank the points in the sector according to the set of random numbers and relabel each point with its rank. Repeat this process for each sector.

4. Form composite number 1 by compositing specimens with the ‹1’ label, form composite number 2 by compositing specimens with the ‹2’ label, etc. This leads to N composite samples that are subjected to chemical analysis.

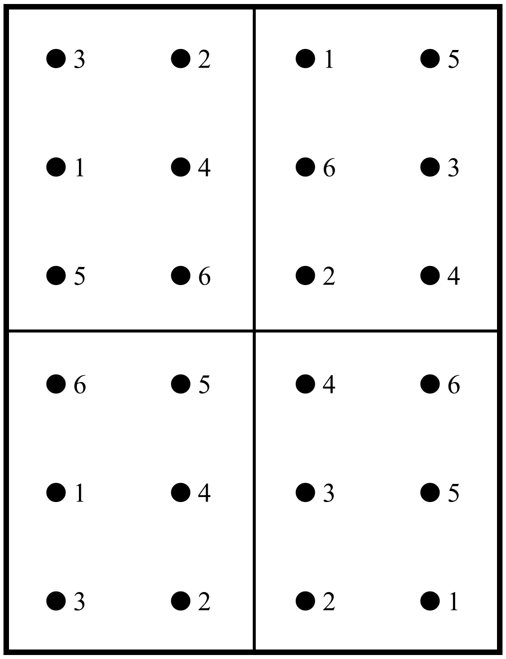


Figure 7. Systematic (square grid points) sample with random compositing scheme (6 composite samples consisting of 4 specimens).

Highlight 5: Procedure for Compositing of Specimens from a Stratified Random Sample Using a Random Scheme (Figure 8)

1. Divide the EA into C sectors (strata) of equal area, where C is equal to the number of specimens to be in each composite (since one specimen per stratum will be used in each composite).

2. Within each stratum, choose N random locations, where N is the desired number of composites. Step-by-step procedures for choosing random locations can be found in Chapter 5 of U.S. EPA (1989a).

3. Use a random number table or random number generator to establish a set of labels for the N points within each sector. This is done by first labeling the points in a sector in an arbitrary fashion (say, points A, B, C,...) and associating the first random number with point A, the second with point B, etc. Then rank the points in the sector according to the set of random numbers and relabel each point with its rank. Repeat this process for each sector.

4. Form composite number 1 by compositing specimens with the '1' label, form composite number 2 by compositing specimens with the '2' label, etc. This leads to N composite samples that are subjected to chemical analysis.

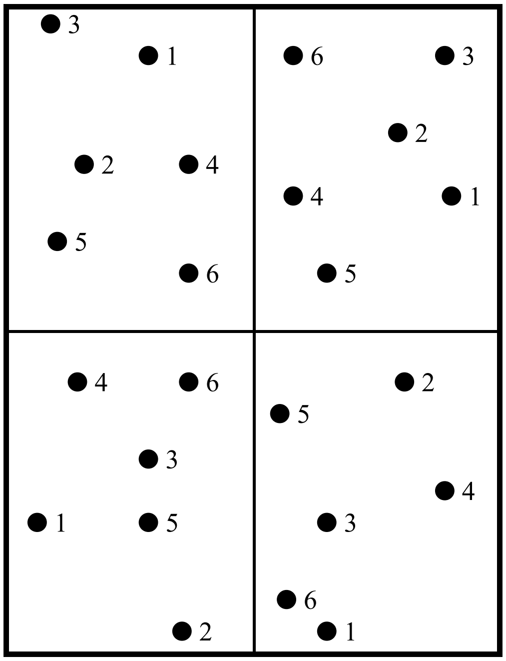


Figure 8. Stratified random sample with random compositing scheme (6 composite samples consisting of 4 specimens).

The error rates shown in Table 4.3 are based on the simulations presented in Appendix I. These simulations are based on the following assumptions:

1. Each of the N composite samples is based on C specimens selected to be representative of the EA as a whole, as specified above (C = number of sectors or strata).

2. One-half the EA has concentrations below the quantitation limit (which is assumed to be SSL/100).

3. One-half the EA has concentrations that follow a gamma distribution (see Section 4.3 for additional discussion).

4. Each chemical analysis is subject to a 20 percent measurement error.

The error rates presented in Table 4.3 are based on the above assumptions which make them robust for most potential distributions of soil radionuclide concentrations. Distribution assumptions 2 and 3 were used because they were found in the simulations to produce high error rates relative to other potential radionuclide distributions (see Section 3.3). If the proportion of the site below the quantitation limit (QL) is less than half or if the distribution of the concentration measurements is some other distribution skewed to the right (e.g., lognormal), rather than gamma, then the error rates achieved are likely to be no worse than those cited in Table 4.3. Although the actual radionuclide distribution may be different from those cited above as the basis for Table 4.3, only extensive investigations will usually generate sufficient data to determine the actual distribution for each EA.

Using Table 4.3 to determine the sample size pair (C and N) needed to achieve satisfactory error rates with the Max test requires an *a priori* estimate of the coefficient of variation for measurements of the radionuclide of interest across the EA. The coefficient of variation (CV) is the ratio of the standard deviation of radionuclide concentrations for individual, uncomposited specimens divided by the EA mean concentration. As discussed in Section 4.1.4, the EAs should be constructed within strata expected to have relatively homogeneous concentrations so that an estimate of the CV for a stratum may be applicable for all EAs in that stratum. The site manager should use a conservatively large estimate of the CV for determining sample size requirements because additional sampling will be needed if the data suggest that the true CV is greater than that used to determine the sample sizes.

Potential sources of information for estimating the EA or stratum means, variances, and CVs include the following (in descending order of desirability):

- Data from a pilot study conducted at the site
- Prior sampling data from the site
- Data from similar sites
- Professional judgment.

For more information on estimating variability, see Section 4.3.1 of U.S. EPA (1989a).

4.1.8 Using the DQA Process: Analyzing Max Test Data. This section provides guidance for analyzing the data for the Max test.

The hypothesis test for the Max test is very simple to implement, which is one reason that the Max test is attractive as a surface soil screening test. If $x_1, x_2, ..., x_N$ represent concentration measurements for N composite samples that each consist of C specimens selected so that each composite is representative of the EA as a whole (as described in Section 4.1.7), the Max test is implemented as follows:

If Max $(x_1, x_2, ..., x_N) \geq 2$ SSL, then investigate the EA further;

If Max (x_1, x_2, ..., x_N) < 2 SSL, and the data quality assessment (DQA) indicates that the sample size was adequate, then no further investigation is necessary.

In addition, the step-by-step procedures presented in Highlight 6 must be implemented to ensure that the site-specific error limits, as discussed in Section 4 .1.6, are achieved.

If the EA mean is below 2 SSL, the DQA process may be used to determine if the sample size was sufficiently large to justify the decision to not investigate further. To use Table 4.3 to check whether the sample size is adequate, an estimate of the CV is needed for each EA. The first four steps of Highlight 6, the DQA process for the Max test, present a process for the computation of a sample CV for an EA based on the N composite samples that each consist of C specimens.

However, the sample CV can be quite large when all the measurements are very small (e.g., well below the SSL) because CV approaches infinity as the EA sample mean ($\bar{x}$) approaches zero. Thus, when the composite concentration values for an EA are all near zero, the sample CV may be questionable and therefore unreliable for determining if the original sample size was sufficient (i.e., it could lead to further sampling when the EA mean is well below 2 SSL). To protect against unnecessary additional sampling in such cases, compare all composites against the equation given in Step 5 of Highlight 6. If the maximum composite sample concentration is below the value given by the equation, then the sample size may be assumed to be adequate and no further DQA is necessary.

To develop Step 5, EPA decided that if there were no compositing (C=1) and all the observations (based on a sample size appropriate for a CV of 2.5) were less than the SSL, then one can reasonably assume that the EA mean was not greater than 2 SSL. Likewise, because the standard error for the mean of C specimens, as represented by the composite sample, is proportional to $1/\sqrt{C}$, the comparable condition for composite observations is that one can reasonably assume that the EA mean was not greater than 2 SSL when all composite observations were less than SSL/$\sqrt{C}$. If this is the case for an EA sample set, the sample size can be assumed to be adequate and no further DQA is needed. Otherwise (when at lease one composite observation is not this small), use Table 4.3 with the sample CV for the EA to determine whether a sufficient number of samples were taken to achieve DQOs.

In addition to being simple to implement, the Max test is recommended because it provides good control over the Type I error rates at 2 SSL with small sample sizes. It also does not need any assumptions regarding observations below the QL. Moreover, the Max test error rates at 2 SSL are fairly robust against alternative assumptions regarding the distribution of surface soil concentrations in the EA. The simulations in Appendix I show that these error rates are rather stable for lognormal or Weibull radionuclide concentration distributions and for different assumptions about portions of the site with radionuclide concentrations below the MDC.

Highlight 6: Directions for Data Quality Assessment for the Max Test

Let $x_1, x_2, ..., x_N$ represent contaminant concentration measurements for N composite samples that each consist of C specimens selected so that each composite is representative of the EA as a whole. The following describes the steps required to ensure that the Max test achieves the DQOs established for the site.

STEP 1: The site manager determines the Type I error rate to be achieved at 2 SSL and the Type II error rate to be achieved at 0.5 SSL, as described in Section 4.1.6.

STEP 2: Calculate the sample mean $\bar{x} = \frac{1}{N} \sum_{i=1}^{N} x_i$

STEP 3: Calculate the sample standard deviation

$$s = \sqrt{\frac{1}{N-1} \sum_{i=1}^{N} (x_i - \bar{x})^2}$$

STEP 4: Calculate the sample estimate of the coefficient of variation,CV, for individual concentration measurements from across the EA.

$$CV = \frac{\sqrt{C}\, s}{\bar{x}}$$

NOTE: This is a conservative approximation of the CV for individual assessments.

STEP 5: If Max $(x_1, x_2, ..., x_N) < \frac{SSL}{\sqrt{C}}$, then no further data quality assessment is needed and the EA needs no further investigation.

Otherwise proceed to Step 6.

STEP 6: Use the value of the sample CV calculated in Step 4 as the true CV of concentrations to determine which column of Table 4.3 is applicable for determining sample size requirements. Using the error limits established in Step 1, determine the sample size requirements from this table. If the required sample size is greater than that implemented, further investigation of the EA is necessary. The further investigation may consist of selecting a supplemental sample and repeating the Max test with the larger, combined sample.

A limitation of the Max test is that it does not provide as good control over the Type II error rates at 0.5 SSL as it does for Type I error rates at 2 SSL. In fact, for a fixed number, C, of specimens per composite, the Type II error rate increases as the number of composite samples, N, increases. As the sample size increases, the likelihood of observing an unusual sample with the maximum exceeding 2 SSL increases. However, the Type II error rate can be decreased by increasing the number of specimens per composite. This unusual performance of the Max test as a hypothesis testing procedure occurs because the rejection region is fixed below 2 SSL and thus does not depend on the sample size (as it does for typical hypothesis testing procedures).

4.1.9 Specify Limits on Decision Errors for Sign Test. Although the Max test is adequate and appropriate for selecting a sample size for site screening, there are other alternate methods of screening surface soils. One such alternate method is the Sign test. While the Sign test is a more complex statistical method than the Max test, it is based on the same statistical null hypothesis (i.e., the EA requires further investigation). Some EAs that cannot be screened out with the Max test could be screened out with the Sign test since it uses a less conservative estimate of the mean concentration than does the Max test.

A disadvantage of the Max test is its performance when the true EA mean is between 0.5 SSL and the SSL. In this case, as the sample size increases, the test indicates the decision to investigate further, even though the mean is less than the SSL. In fact, no test procedure with feasible sample sizes performs well when the true EA mean is in the "gray region" between 0.5 SSL and 2 SSL (see Section 4.3).

However, improved performance may be achieved in this region through the use of nonparametric statistical tests. The basic difference between parametric and nonparametric statistical techniques is that parametric techniques, such as the Max test, use specific assumptions about the probability distributions of the measurement data. Non parametric techniques (sometimes referred to as distribution-free statistical methods) can be used without assuming a particular underlying distribution. Thus, nonparametric techniques are often more appropriate in situations when the probability distribution of the data is either unknown or is some continuous distribution other than the normal distribution.

For radiological measurements at levels associated with SSLs, the data are likely to approach the MDC for typical radioanalytical methods. There may, therefore, be some measurement data which are at or below MDC, and the variability of the measurements will be relatively large. Such data are not easily treated using parametric methods, thus offering an additional advantage to the use of nonparametric techniques.

The Sign test is a nonparametric statistical test recommended for use as an alternative to the Max test. As is the case for many nonparametric tests, it is based on ranking the measurement data. The data are ordered from smallest to largest, and assigned numbers (ranks) 1, 2, 3, ... accordingly. The analysis is then performed on the ranks rather than on the original measurement values. The advantage of this approach is that the probability that one measurement is larger (i.e., ranked higher) than another can be computed exactly by combinatorial methods without reference to a specific probability distribution.

The Soil Screening Guidance for chemicals proposes the use of the Chen test as an alternative to the Max test. This requires the use of a different null hypothesis and different decision error rates than those for the Max test. An additional advantage of using the Sign test as the alternative to the Max test is that the same null hypothesis and decision error rates are used for both tests. Therefore, the null hypothesis for the Sign test is the same as given in Section 4.1.6, namely that the site needs further investigation. The Type I and Type II decision error rates are the same as those given in Section 4.1.6:

- Prob (“walk away” when the true EA mean is 2 SSL) = 0.05
- Prob (“investigate further” when the true EA mean is 0.05 SSL) = 0.20.

The DQO Process for the Sign test is therefore the same as that given for the Max test and is shown in Table 4.2.

4.1.10 Optimize the Design Using the Sign Test. This section includes guidance on developing an optimum sampling strategy for screening surface soils. It discusses compositing, the selection of sampling points for composited and uncomposited surface soil sampling, and the recommended procedures for determining the sample sizes necessary to achieve specified limits on decision errors using the Sign test.

The general guidance provided in Section 4.1.7 for optimizing the design for the Max test is also applicable for the Sign test.

Compositing. The guidance provided in Section 4.1.7 for compositing samples for the Max test is also applicable for the Sign test.

Sample Pattern. The Sign test can be applied using composite samples that are representative of the entire EA or with individual uncomposited samples.

The guidance provided in Section 4.1.7 concerning sample patterns and compositing schemes for the Max test are also applicable to the Sign test.

Sample Size. This section provides procedures to determine sample size requirements for the Sign test that achieve the site-specific decision error limits discussed in Section 4 .1.6. **Calculate the Relative Shift.** The initial step in determining the number of data points for the Sign test is to calculate a term known as the relative shift, Δ/σ_s. This is defined as the ratio of the width of the gray region, Δ, and the standard deviation of the radionuclide in the EA, σ_s. As specified in Section 4.1.6, the gray region ranges from 0.5 SSL to 2.0 SSL, therefore the width of the gray region is 1.5 SSL. Also as described in Section 4.1.7, the value of σ_s may be obtained from data from a pilot study conducted at the site, prior sampling data from the site, data from similar sites, or professional judgement. Values of the relative shift that are less than one will result in a large number of measurements needed to achieve the site-specific error limits discussed in Section 4.1.6.

The importance of choosing an appropriate value for σ_s must be emphasized. If the value is grossly underestimated, the number of data points will be too few to obtain the desired power level for the test and a resurvey may be recommended. If, on the other hand, the value is overestimated, the number of data points determined will be unnecessarily large.

Determine Sign p. Sign p is the estimated probability that a random measurement from the EA will be less than the upper bound of the gray region (i.e., 2.0 SSL) when the EA median is actually at the lower bound of the gray region (i.e., 0.5 SSL). The Sign p is used to calculate the minimum number of data points necessary for the survey to meet the DQOs. The value of the relative shift calculated in the previous section is used to obtain the corresponding value of Sign p from Table 4.4.

Table 4.4. Values of Sign p for Given Values of the Relative Shift, Δ/σ

Δ/σ	Sign p	Δ/σ	Sign p
0.1	0.539828	1.2	0.884930
0.2	0.579260	1.3	0.903199
0.3	0.617911	1.4	0.919243
0.4	0.655422	1.5	0.933193
0.5	0.691462	1.6	0.945201
0.6	0.725747	1.7	0.955435
0.7	0.758036	1.8	0.964070
0.8	0.788145	1.9	0.971284
0.9	0.815940	2.0	0.977250
1.0	0.841345	2.5	0.993790
1.1	0.864334	3.0	0.998650

If $\Delta/\sigma > 3.0$, use Sign p = 1.000000

Determine Decision Error Percentiles. The next step in this process is to determine the percentiles, $Z_{1-\alpha}$ and $Z_{1-ß}$, represented by the selected decision error levels, α and ß, respectively (see Table 4.5).

Table 4.5. Percentiles Represented by Selected Values of α and ß

α (or ß)	$Z_{1-\alpha}$ (or $Z_{1-ß}$)	α (or β)	$Z_{1-\alpha}$ (or $Z_{1-\beta}$)
0.005	2.576	0.10	1.282
0.01	2.326	0.15	1.036
0.015	2.241	0.20	0.842
0.025	1.960	0.25	0.674
0.05	1.645	0.30	0.524

Calculate Number of Data Points for Sign Test. The number of data points, N, to be obtained for the Sign test is next calculated using the following formula:

$$N = \frac{(Z_{1-\alpha} + Z_{1-\beta})^2}{4\,(Sign\ P - 0.5)^2} \tag{36}$$

The value of N calculated using this equation is an approximation based on estimates of σ, so there is some uncertainty associated with this calculation. In addition, there will be some missing or unusable data from any sampling and analysis program. The rate of missing or unusable measurements expected to occur in an EA and the uncertainty associated with the calculation of N should be accounted for during the sampling and analysis program planning stage. The number of data points should be increased by 20%, and rounded up, over the values calculated using the above equation to obtain sufficient data points to attain the desired power level with the statistical tests and allow for possible lost or unusable data. The value of 20% is selected to account for a reasonable amount of uncertainty in the parameters used to calculate N and still allow flexibility to account for some lost or unusable data. The recommended 20% correction factor should be applied as a minimum value. Experience and site-specific considerations should be used to increase the correction factor if required. If the user determines that the 20% increase in the number of measurements is excessive for a specific site, a retrospective power curve should be used to demonstrate that the survey design provides adequate power to support the decision.

Obtain Number of Data Points for Sign Test from Table 4.6. Table 4.6 provides a list of the number of data points necessary to achieve the desired Type I error rate of 0.05 and Type II error rate of 0.20 as a function of the relative shift, Δ/σ. The values listed in Table 4.6 represent the number of measurements to be performed in each EA. These values were calculated using above equation for N and increased by 20% to account for missing or unusable data and uncertainty in the calculated value of N.

Table 4.6. Values of N for a Given Relative Shift (Δ/σ) for α = 0.05 and β = 0.20

Δ/σ	N
0.1	1170
0.2	296
0.3	134
0.4	77
0.5	51
0.6	37
0.7	28
0.8	23
0.9	19
1.0	16
1.2	13
1.4	11
1.6	10
1.8	9
2.0	9
2.5	8
3.0	8

Given an *a priori* estimate of the standard deviation, σ_s of concentration measurements in the EA, the site manager can use Table 4.6 to determine a sample size option that achieves the decision error goals for surface soil screening presented in Section 4 .1.6 (i.e., not more than 20 percent chance of error at 0.5 SSL and not more than 5 percent at 2 SSL). For example, suppose that the site manager expects that the ratio of the width of the gray region and the maximum true σ_s for concentration measurements in an EA is 2. Then Table 4.6 shows that a total of nine samples will be sufficient to achieve the decision error limit goals.

4.1.11 Using the DQA Process: Analyzing Sign Test Data.

Data Quality Assessment (DQA) is a scientific and statistical evaluation that determines if the data are of the right type, quality, and quantity to support their intended use. Prior to conducting the Sign test on the measurement data, a preliminary data review should be conducted.

Preliminary Data Review. The purpose of this phase of the DQA process is to conduct a preliminary evaluation of the data set to determine that the data are consistent with the underlying assumptions upon which the statistical test is based. In order to learn about the structure of the data—identifying patterns, relationships, or potential anomalies—basic statistical quantities (*i.e.*, mean, standard deviation, median) should be calculated and the data should be plotted (i.e., a frequency plot and a posting plot). In addition, a retrospective power curve should be constructed to examine if the DQOs have been achieved.

Basic Statistical Quantities

- The mean of the data. The mean of the data for the EA can be compared to 2 times the SSL to get a preliminary indication of the EA status. If the site is indeed contaminated, this comparison may readily reveal that a EA contains excess residual radioactivity—even before applying statistical tests. For example, if the EA average concentration exceeds 2 times the SSL, then the EA clearly does not meet the screening criteria. On the other hand, if the maximum EA measurement is less than 2 times the SSL, the EA meets the screening criteria. Likewise, if every measurement in the EA is less than 2 times the SSL, the EA meets the screening criteria and no statistical tests are required.

- The sample standard deviation. The value of the sample standard deviation is especially important. If too large compared to that assumed during the survey design, this may indicate an insufficient number of samples were collected to achieve the desired power of the statistical test. Again, inadequate power can lead to unnecessary additional surveys.

- The median of the data. The median is the middle value of the data set when the number of data points is odd, and is the average of the two middle values when the number of data points is even. Thus, 50% of the data points are above the median, and 50% are below the median. Large differences between the mean and the median would be an early indication of skewness in the data. This would also be evident in a histogram of the data. For example, if the difference between the median and the mean is a small fraction of the sample standard deviation, the mean and median would not be considered significantly different.

- Minimum, maximum, and range of data. Examining the minimum, maximum, and range of the data may provide additional useful information. If the difference between the minimum and the maximum is a small number of standard deviations, the overall range is not unusually large. When there are 30 or fewer data points, values of the range much larger than about 4 to 5 standard deviations would be unusual. For larger data sets the range might be wider.

Graphical Data Review

- Posing plot. Spatial dependencies that potentially affect the assumptions can be assessed using posting plots. A posting plot is simply a map of the EA with the data values entered at the measurement locations. This potentially reveals heterogeneities in the data—especially possible small areas of elevated activity.

  For example, a plot may reveal an obvious trend towards larger values as one moves from left to right across the EA. This trend may not be apparent from a simple listing of the data. The trend may become even more apparent if isopleths are added to the posting plot. If the posting plot reveals systematic spatial trends in the EA, the cause of the trends would need to be investigated. More sophisticated tools for determining the extent of spatial dependencies are also available, such as the *Guidance for Data Quality Assessment: Practical Methods for Data Analysis*, EPA QA/G-9 (*e.g.*, U.S. EPA, 1996c). These methods tend to be complex and are best used with guidance from a professional statistician.

- Frequency plot. A frequency plot (or a histogram) is a useful tool for examining the general shape of a data distribution. This plot is a bar chart of the number of data points within a certain range of values. A simple method for generating a rough frequency plot is the stem and leaf display. The frequency plot will reveal any obvious departures from symmetry, such as skewness or bimodality (two peaks), in the data distributions for the EA. Skewness or other asymmetry can impact the accuracy of the statistical tests. A data transformation (*e.g.*, taking the logarithms of the data) can sometimes be used to make the distribution more symmetric. The statistical tests would then be performed on the transformed data. When the underlying data distribution is highly skewed, it is often because there are a few high areas.

  The presence of two peaks in the EA frequency plot may indicate the existence of isolated areas of residual radioactivity. It may also indicate a mixture of soil types, construction materials, *etc.* The greater variability in the data due to the presence of such a mixture will reduce the power of the statistical tests to detect an adequately remediated EA. These situations should be avoided whenever possible by carefully choosing EAs.

- Quantile plot. A Quantile plot is constructed by first ranking the data from smallest to largest. Sorting the data is easy once the stem and leaf display has been constructed. Then, each data value is simply plotted against the percentage of the samples with that value or less. The slope of the curve in the Quantile plot is an indication of the amount of data in a given range of values. A small amount of data in a range will result in a large slope. A large amount of data in a range of values will result in a more horizontal slope. A sharp rise near the bottom or the top is an indication of asymmetry. Sudden changes in slope, or notably steep areas may indicate peculiarities in the data needing further investigation.

MARSSIM contains a detailed discussion of graphical data review methods. In addition, *Guidance for Data Quality Assessment: Practical Methods for Data Analysis*, EPA QA/G-9 (*e.g.*, U.S. EPA, 1996c) provides additional information concerning interpretation of these methods.

Retrospective Power Curve

One of the more important assumptions made in the survey design described earlier is that the sample sizes determined for the tests must be sufficient to achieve the data quality objectives set for the Type I (α) and Type II (β) error rates. A retrospective power curve should be constructed using the actual number of measurements obtained, N, and the measured value of σ. The resulting power curve shows the probability that the EA could be screened out using the Sign test versus the screening level. This curve can show if the DQOs have been achieved. The consequence of inadequate power is that a EA that should be screened out has a higher probability of being incorrectly deemed to require further investigation.

An effective sampling and analysis plan will slightly overestimate both the number of measurements and the standard deviation to ensure adequate power. This insures that an EA is not subjected to additional remediation simply because the measurements is not sensitive enough to detect that residual radioactivity is below the guideline level. When the null hypothesis is rejected, the power of the test becomes a somewhat moot question. Nonetheless, even in this case, a retrospective power curve can be a useful diagnostic tool and an aid to designing future surveys.

Analyze the Data Using the Sign Test. Step-by-step instructions for using the Sign test to analyze data from both discrete random samples and pseudo-random samples (e.g., composite samples constructed as described previously) are provided in Highlight 7.

Highlight 7: Directions for the Sign Test

STEP 1: List the EA measurements, $x_1, x_2, ..., x_N$.

STEP 2: Subtract each measurement, x_i , from the upper bound of the gray area, 2 SSL to obtain the differences:

$$D_i = 2\ SSL - x_i \qquad i = 1, 2, ..., N.$$

STEP 3: Discard each difference that is exactly zero and reduce the sample size, *N*, by the number of such zero measurements.

STEP 4: Count the number of positive differences. The result is the Sign test statistic *S*+. Note that a positive difference corresponds to a measurement below 2 SSL and contributes evidence that the EA does not require further investigation. The larger the number of positive differences, the more likely that the null hypothesis may be rejected.

STEP 5: The value of *S*+ is compared to the critical value for the Sign test, k_{Sign} , given in Table 4.7, corresponding to the desired Type I error rate, α, and the number of measurements N. If *S*+ is greater than the critical value, k_{Sign}, in Table 5.7, the null hypothesis is rejected and the EA does not require further investigation.

Table 4.7. Critical Values k_{Sign} for the Sign Test Statistic S+

N	**Type I error rate, α**								
	0.005	**0.01**	**0.025**	**0.05**	**0.1**	**0.2**	**0.3**	**0.4**	**0.5**
4	4	4	4	**4**	3	3	3	2	2
5	5	5	5	**4**	4	3	3	3	2
6	6	6	5	**5**	5	4	4	3	3
7	7	6	6	**6**	5	5	4	4	3
8	7	7	7	**6**	6	5	5	4	4
9	8	8	7	**7**	6	6	5	5	4
10	9	9	8	**8**	7	6	6	5	5
11	10	9	9	**8**	8	7	6	6	5
12	10	10	9	**9**	8	7	7	6	6
13	11	11	10	**9**	9	8	7	7	6
14	12	11	11	**10**	9	9	8	7	7
15	12	12	11	**11**	10	9	9	8	7
16	13	13	12	**11**	11	10	9	9	8
17	14	13	12	**12**	11	10	10	9	8
18	14	14	13	**12**	12	11	10	10	9
19	15	14	14	**13**	12	11	11	10	9
20	16	15	14	**14**	13	12	11	11	10
21	16	16	15	**14**	13	12	12	11	10

Table 4.7. Critical Values k_{Sign} for the Sign Test Statistic S+

	Type I error rate, α								
N	**0.005**	**0.01**	**0.025**	**0.05**	**0.1**	**0.2**	**0.3**	**0.4**	**0.5**
22	17	16	16	**15**	14	13	12	12	11
23	18	17	16	**15**	15	14	13	12	11
24	18	18	17	**16**	15	14	13	13	12
25	19	18	17	**17**	16	15	14	13	12
26	19	19	18	**17**	16	15	14	14	13
27	20	19	19	**18**	17	16	15	14	13
28	21	20	19	**18**	17	16	15	15	14
29	21	21	20	**19**	18	17	16	15	14
30	22	21	20	**19**	19	17	16	16	15
31	23	22	21	**20**	19	18	17	16	15
32	23	23	22	**21**	20	18	17	17	16
33	24	23	22	**21**	20	19	18	17	16
34	24	24	23	**22**	21	19	19	18	17
35	25	24	23	**22**	21	20	19	18	17
36	26	25	24	**23**	22	21	20	19	18
37	26	26	24	**23**	22	21	20	19	18
38	27	26	25	**24**	23	22	21	20	19
39	27	27	26	**25**	23	22	21	20	19
40	28	27	26	**25**	24	23	22	21	20
41	29	28	27	**26**	25	23	22	21	20
42	29	28	27	**26**	25	24	23	22	21
43	30	29	28	**27**	26	24	23	22	21
44	30	30	28	**27**	26	25	24	23	22
45	31	30	29	**28**	27	25	24	23	22
46	32	31	30	**29**	27	26	25	24	23
47	32	31	30	**29**	28	26	25	24	23
48	33	32	31	**30**	28	27	26	25	24
49	33	33	31	**30**	29	27	26	25	24
50	34	33	32	**31**	30	28	27	26	25

A retrospective power analysis for the test will often be useful, especially when the null hypothesis is not rejected (see Appendix I.9). When the null hypothesis is not rejected, it may be because it is in fact true, or it may be because the test did not have sufficient power to detect that it is not true. The power of the test will be primarily affected by changes in the actual number of measurements obtained and their standard deviation. A retrospective power curve should be constructed using the actual number of measurements obtained, N, and the measured value of σ. The resulting power curve shows the probability that the EA could be screened out using the Sign test. This curve can show if the DQOs have been achieved. The consequence of inadequate power is that an EA that should be screened out has a higher probability of being incorrectly deemed to require further investigation.

An effective sampling and analysis plan will slightly overestimate both the number of measurements and the standard deviation to ensure adequate power. This insures that an EA is not subjected to additional remediation simply because the measurements is not sensitive enough to detect that residual radioactivity is below the guideline level. When the null hypothesis is rejected, the power of the test becomes a somewhat moot question. Nonetheless, even in this case, a retrospective power curve can be a useful diagnostic tool and an aid to designing future surveys.

4.1.12 Special Considerations for Multiple Radionuclides. If the surface soil samples collected for an EA will be tested for multiple radionuclides , be aware that the expected CVs or σ for the different radionuclides may not all be identical. A conservative approach is to base the sample sizes for all radionuclides on the largest expected CV or σ.

4.1.13 Quality Assurance/Quality Control Requirements. Regardless of the sampling approach used, the Superfund quality assurance program guidance should be followed to ensure that measurement error rates are documented and within acceptable limits (U.S. EPA, 1993d).

4.1.14 Final Analysis. After either the Max test or the Sign test has been performed for each EA of interest (0.5 acre or less) at an NPL site, the pattern of decisions for individual EAs (to "walk away" or to "investigate further") should be examined. If some EAs for which the decision was to "walk away" are surrounded by EAs for which the decision was to "investigate further," it may be more efficient to identify an area including all these EAs for further study and develop a global investigation strategy.

4.1.15 Reporting. The decision process for surface soil screening should be thoroughly documented as part of the RI/FS process. This documentation should include a map of the site (showing the boundaries of the EAs and the sectors, or strata, within EAs that were used to select sampling points within the EAs); documentation of how composite samples were formed and the number of composite samples that were analyzed for each EA; the raw analytical data; the results of all hypothesis tests; and the results of all QA/QC analyses.

4.2 Sampling Subsurface Soils

Subsurface soil sampling is conducted to estimate the mean concentrations of radionuclides in each source at a site for comparison to inhalation and migration to ground water SSLs. Measurements of soil properties and estimates of the area and depth of contamination in each source are also needed to calculate SSLs for these pathways. Table 4.8 shows the steps in the DQO process necessary to develop a sampling strategy to meet these objectives. Each of these steps is described below.

Table 4.8. Soil Screening DQOs for Subsurface Soils

DQO Process Steps	Soil Screening Inputs/Outputs
State the Problem	
Identify scoping team	Site manager and technical experts (e.g., health physicists, risk assessors, hydrogeologists, statisticians).
Develop conceptual site model (CSM)	CSM development (described in Step 1 of the User's Guide, U.S. EPA, 1999).
Define exposure scenarios	migration of radionuclides from soil to potable ground water.
Specify available resources	Sampling and analysis budget, scheduling constraints, and available personnel.
Write brief summary of contamination problem	Summary of the subsurface soil contamination problem to be investigated at the site.
Identify the Decision	
Identify decision	Do mean soil concentrations for particular radionuclides (e.g., radionuclides of potential concern) exceed appropriate SSLs?
Identify alternative actions	Eliminate area from further action or study under CERCLA or Plan and conduct further investigation.
Identify Inputs to the Decision	
Identify decision	migration to ground water SSLs for specified radionuclides Measurements of subsurface soil radionuclide concentration
Define basis for screening	Soil Screening Guidance for Radionuclides
Identify analytical methods	Feasible analytical methods (both field and laboratory) consistent with program-level requirements.
Specify the Study Boundaries	
Define geographic areas of field investigation	The entire NPL site (which may include areas beyond facility boundaries), except for any areas with clear evidence that no contamination has occurred.
Define population of interest	Subsurface soils
Define scale of decision making	Sources (areas of contiguous soil contamination, defined by the area and depth of contamination or to the water table, whichever is more shallow).
Subdivide site into decision units	Individual sources delineated (area and depth) using existing information or field measurements (several nearby sources may be combined into a single source).
Define temporal boundaries of study	Temporal constraints on scheduling field visits.
Identify (list) practical constraints	Potential impediments to sample collection, such as access, health, and safety issues.
Develop a Decision Rule	
Specify parameter of interest	Mean soil radionuclide concentration in a source (as represented by discrete radionuclide concentrations averaged within soil borings).
Specify screening level	SSLs calculated using available parameters and site data (or generic SSLs if site data are unavailable).
Specify ?if..., then..." decision rule	If the mean soil concentration exceeds the SSL, then investigate the source further. If the mean soil boring concentration is less than the SSL, then no further investigation is required under CERCLA.

Table 4.8. Soil Screening DQOs for Subsurface Soils

DQO Process Steps	Soil Screening Inputs/Outputs
Specify Limits on Decision Errors	
Define QA/QC goals	Radioanalytical laboratory precision and bias requirements 10% laboratory analyses for field methods
Optimize the Design	
Determine how to estimate mean concentration in a source	For each source, the highest mean soil core concentration (i.e., depth-weighted average of discrete radionuclide concentrations within a boring).
Define subsurface sampling strategy by evaluating costs and site-specific conditions	Number of soil borings per source area; number of sampling intervals with depth.
Develop planning documents for the field investigation	Sampling and Analysis Plan (SAP) Quality Assurance Project Plan (QAPjP)

4.2.1 State the Problem. Radionuclides present in subsurface soils at the site may pose significant risk to human health and the environment by the migration of radionuclides through soils to an underlying potable aquifer. The problem is to identify the radionuclides and source areas that do not pose significant risk to human health so that future investigations may be focused on areas and radionuclides of true concern.

Site-specific activities in this step include identifying the data collection planning team (including technical experts and key stakeholders) and specifying the available resources (i.e., the cost and time available for sampling). The list of technical experts and stakeholders should contain all key personnel who are involved with applying SSLs to the site. Other activities include developing the conceptual site model and identifying exposure scenarios, which are fully addressed in the *Soil Screening Guidance for Radionuclides: User's Guide* (U.S. EPA, 2000).

4.2.2 Identify the Decision. The decision is to determine whether mean soil concentrations in each source area exceed migration to ground water SSLs for specific radionuclides. If so, the source area will be investigated further. If not, no further action will be taken under CERCLA.

4.2.3 Identify Inputs to the Decision. Site-specific inputs to the decision include the average radionuclide concentrations within each source area and the inhalation and migration ground water SSLs. Calculation of the SSLs for the two pathways of concern also requires site-specific measurements of soil properties (i.e., bulk density, pH, and soil texture class) and estimates of the areal extent and depth of contamination.

A list of feasible sampling and analytical methods should be assembled during this step. EPA recommends the use of field methods where applicable and appropriate. Verify that radioanalytical laboratory methods and field methods for analyzing the samples exist and that the analytical method detection limits or field method detection limits are appropriate for the site-specific or generic SSL. MARSSIM (U.S. EPA, 1997c) contains further information concerning the collection and preparation of samples for analysis as well as a general discussion of radiological laboratory sample preparation and analysis methods, and in evaluating the results of these analyses.

Field methods will be useful in defining the study boundaries (i.e., area and depth of contamination) during site reconnaissance and during the sampling effort. For radioactive contaminants, core samples may also be and monitored intact in the field to determine if layers of radioactivity are present. In addition, the use of a subsurface sampling technique, which results in a borehole or soil face, may be logged using a gamma scintillation detector. This enables scanning of the exposed soil surface to identify radioactive contamination within small fractions of hole depth, thus facilitating the identification of the presence and depth distribution of subsurface radioactivity. This information may be used to direct further core sampling and laboratory analysis as warranted. EPA expects field methods will become more prevalent and useful because the design and capabilities of field portable instrumentation are rapidly evolving. Documents on standard operating procedures (SOPs) for field methods are available through NTIS and should be referenced in soil screening documentation if these methods are used.

Soil parameters necessary for SSL calculation are soil texture and bulk density. Some of these parameters can be measured in the field, others require laboratory measurement. Although laboratory measurements of these parameters cannot be obtained under the Superfund Contract Laboratory Program, they are readily available from soil testing laboratories across the country.

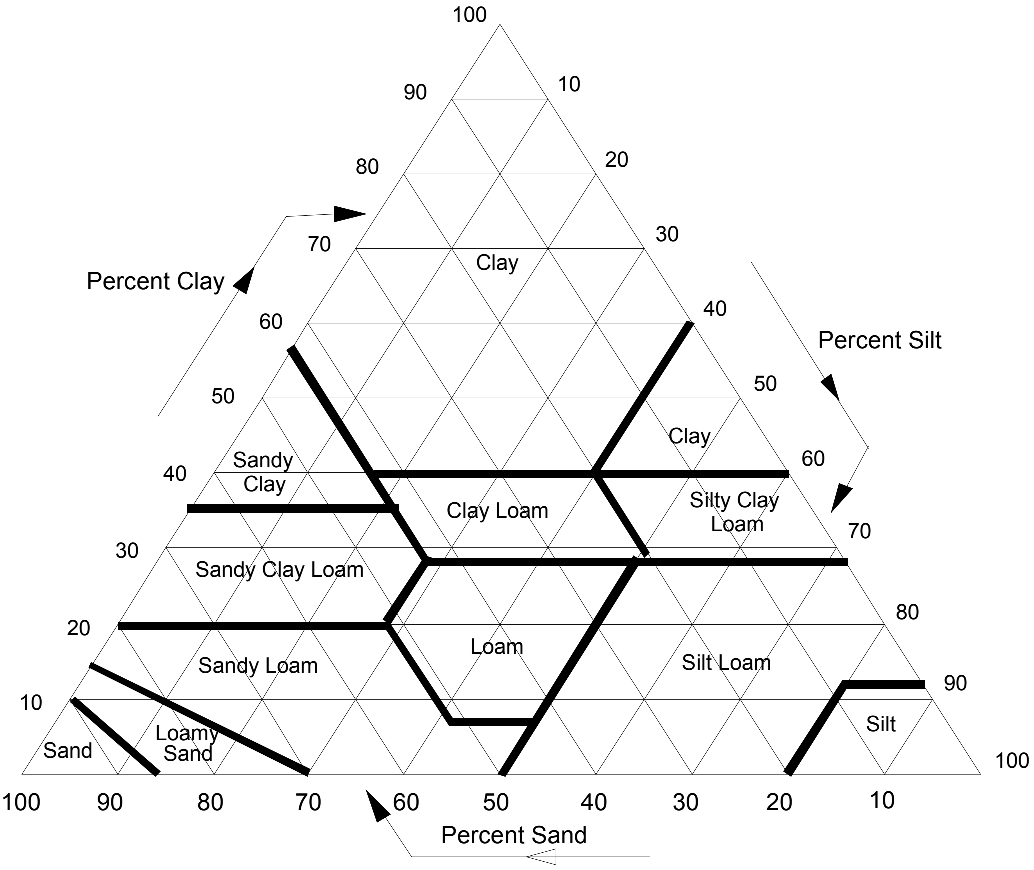
Note that the size, shape, and orientation of sampling volume (i.e., ?support") for heterogenous media have a significant effect on reported measurement values. For instance, particle size has a varying affect on the transport and fate of radionuclides in the environment and on the potential receptors. Comparison of data from methods that are based on different supports can be difficult. Defining the sampling support is important in the early stages of site characterization. This may be accomplished through the DQO process with existing knowledge of the site, contamination, and identification of the exposure pathways that need to be characterized. Refer to *Preparation of Soil Sampling Protocols: Sampling Techniques and Strategies* (U.S. EPA, 1992f) for more information about soil sampling support.

Soil Texture. The soil texture class (e.g., loam, sand, silt loam) is necessary to estimate average soil moisture conditions and to estimate infiltration rates. A soil's texture classification is determined from a particle size analysis and the U.S. Department of Agriculture (USDA) soil textural triangle shown at the top of Figure 9. This classification system is based on the USDA soil particle size classification at the bottom of Figure 9. The particle size analysis method in Gee and Bauder (1986) can provide this particle size distribution also. Other particle size analysis methods may be used as long as they provide the same particle size breakpoints for sand/silt (0.05 mm) and silt/clay (0.002 mm). Field methods are an alternative for determining soil textural class; an example from Brady (1990) is also presented in Figure 9.

Dry Bulk Density. Dry soil bulk density (ρ_b) is used to calculate total soil porosity and can be determined for any soil horizon by weighing a thin-walled tube soil sample (e.g., Shelby tube) of known volume and subtracting the tube weight to estimate field bulk density (ASTM D 2937). A moisture content determination (ASTM 2216) is then made on a subsample of the tube sample to adjust field bulk density to dry bulk density. The other methods (e.g., ASTM D 1556, D 2167, D 2922) are not generally applicable to subsurface soils. ASTM soil testing methods are readily available in the *Annual Book of ASTM Standards, Volume 4.08, Soil and Rock; Building Stones*, which is available from ASTM, 100 Barr Harbor Drive, West Conshohocken, PA, 19428.

pH. Soil pH is used to select site-specific partition coefficients for metals and ionizing organic compounds (see Part 4). This simple measurement is made with a pH meter in a soil/water slurry (McLean, 1982) and may be measured in the field using a portable pH meter.

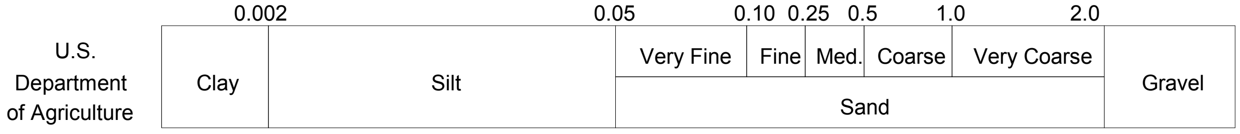
Exhibit 9: U.S. Department of Agriculture soil texture classification



Criteria Used with the Field Method for Determining Soil Texture Classes (Source: Brady, 1990)

Criterion	Sand	Sandy loam	Loam	Silt loam	Clay loam	Clay
1. Individual grains visible to eye	Yes	Yes	Some	Few	No	No
2. Stability of dry clods	Do not form	Do not form	Easily broken	Moderately easily broken	Hard and stable	Very hard and stable
3. Stability of wet clods	Unstable	Slightly stable	Moderately stable	Stable	Very stable	Very stable
4. Stability of "ribbon" when wet soil rubbed between thumb and fingers	Does not form	Does not form	Does not form	Broken appearance	Thin, will break	Very long, flexible

Particle Size, mm



Source: USDA.

4.2.4 Define the Study Boundaries.

As discussed in Section 4.1.4, areas that are known to be highly contaminated (i.e., sources) are targeted for subsurface sampling. The information collected on source area and depth is used to calculate site-specific SSLs for the inhalation and migration to ground water pathways. For the purposes of this guidance, source areas are defined by area and depth as contiguous zones of contamination. However, discrete sources that are near each other may be combined and investigated as a single source if site conditions warrant.

4.2.5 Develop a Decision Rule.

The decision rule for subsurface soils is:

> If the mean concentration of a radionuclide within a source area exceeds the screening level,
> then investigate that area further.

In this case "screening level" means the SSL. As explained in Section 4.1.5, statistics other than the mean (e.g., the maximum concentration) may be used as estimates of the mean in this comparison as long as they represent valid or conservative estimates of the mean.

4.2.6 Specify Limits on Decision Errors.

EPA recognizes that data obtained from sampling and analysis can never be perfectly representative or accurate and that the costs of trying to achieve near-perfect results can outweigh the benefits. Consequently, EPA acknowledges that uncertainty in data must be tolerated to some degree. The DQO process attempts to control the degree to which uncertainty in data affects the outcomes of decisions that are based on data.

The sampling intensity necessary to accurately determine the mean concentration of subsurface soil contamination within a source with a specified level of confidence (e.g., 95 percent) is impracticable for screening due to excessive costs and difficulties with implementation. Therefore, EPA has developed an alternative decision rule based on average concentrations within individual soil cores taken in a source:

> If the mean concentration within **any** soil core taken in a source exceeds the screening level,
> then investigate that source further.

For each core, the mean core concentration is defined as the depth-weighted average concentration within the zone of contamination (see Section 4.2.7). Since the soil cores are taken in the area(s) of highest contamination within each source, the highest average core concentration among a set of core samples serves as a conservative estimate of the mean source concentration. Because this rule is not a statistical decision, it is not possible to statistically define limits on decision errors.

Standard limits on the precision and bias of sampling and analytical operations conducted during the sampling program do apply. These are specified by the Superfund quality assurance program requirements (U.S. EPA, 1993d), which must be followed during the subsurface sampling effort.

If field methods are used, at least 10 percent of field samples should be split and sent to a radioanalytical laboratory for confirmatory analysis (U.S. EPA, 1993d).

Although the EPA does not require full laboratory sample tracking and quality assurance/quality control (QA/QC) procedures for measurement of soil properties, routine EPA QA/QC procedures are recommended, including a Quality Assurance Project Plan (QAPjP), chain-of-custody forms, and duplicate analyses.

4.2.7 Optimize the Design. Within each source, the Soil Screening Guidance for Radionuclides suggests taking two to three soil cores using split spoon or Shelby tube samplers. For each soil core, samples should begin at the ground surface and continue at approximately 2-foot intervals until no contamination is encountered or to the water table, whichever is shallower. **Subsurface sampling depths and intervals can be adjusted at a site to accommodate site-specific information on surface and subsurface radionuclide distributions and geological conditions** (e.g., very deep water table, very thick uncontaminated unsaturated zone, user well far beyond edge of site, soils underlain by karst or fractured rock aquifers).

The number and location of subsurface soil sampling (i.e., soil core) locations should be based on knowledge of likely surface soil contamination patterns and subsurface conditions. This usually means that core samples should be taken directly beneath areas of high surface soil contamination. Surface soils sampling efforts and field measurements (e.g., soil gas surveys) taken during site reconnaissance will provide information on source areas and high radionuclide concentrations to help target subsurface sampling efforts. Information in the CSM also will provide information on areas likely to have the highest levels of contamination. Note that there may be sources buried in subsurface soils that are not discernible at the surface. Information on past practices at the site included in the CSM can help identify such areas. Surface geophysical methods also can aid in identifying such areas (e.g., magnetometry to detect buried drums).

The intensity of the subsurface soil sampling needed to implement the soil screening process typically will not be sufficient to fully characterize the extent of subsurface contamination. In these cases, conservative assumptions should be used to develop hypotheses on likely radionuclide distributions (e.g., the assumption that soil contamination extends to the water table). Along with knowledge of subsurface hydrogeology and stratigraphy, geostatistics can be a useful tool in developing subsurface radionuclide distributions from limited data and can provide information to help guide additional sampling efforts. However, instructions on the use of geostatistics is beyond the scope of this guidance.

Samples for measuring soil parameters should be collected when taking samples for measuring radionuclide concentrations. If possible, consider splitting single samples for radionuclide and soil parameter measurements. A number of soil testing laboratories have provisions in place for handling and testing radioactive samples. However, if testing radioactive samples is a problem, samples may be taken from clean areas of the site as long as they represent the same soil texture and series and are taken from the same depth as the radionuclide concentration samples.

The SAP developed for subsurface soils should specify sampling and analytical procedures as well as the development of QA/QC procedures. To identify the appropriate analytical procedures, the screening levels must be known. If data are not available to calculate site-specific SSLs, then the generic SSLs in Appendix A should be used.

Finally, soil investigation for the migration to ground water pathway should not be conducted independently of ground water investigations. Contaminated ground water may indicate the presence of a nearby source area, with radionuclides leaching from soil into the aquifer.

4.2.8 Analyzing the Data. The mean soil radionuclide concentration for each soil core should be compared to the SSL for the radionuclide. The soil core average should be obtained by averaging analyses results for the discrete samples taken along the entire soil core within the zone of contamination (compositing will prevent the evaluation of radionuclide concentration trends with depth).

If each subsurface soil core segment represents the same subsurface soil interval (e.g., 2 feet), then the average concentration from the surface to the depth of contamination is the simple arithmetic average of the concentrations measured for core samples representative of each of the 2-foot segments from the surface to the depth of contamination or to the water table. However, if the intervals are not all of the same length (e.g., some are 2 feet while others are 1 foot or 6 inches), then the calculation of the average concentration in the total core must account for the different lengths of the intervals.

If c_i is the concentration measured in a core sample representative of a core interval of length l_i, and the n-th interval is considered to be the last interval in the source area (i.e., the n-th sample represents the depth of contamination), then the average concentration in the core from the surface to the depth of contamination should be calculated as the following depth-weighted average ($\bar{c}$),

$$\bar{c} = \frac{\sum_{i=1}^{n} l_i c_i}{\sum_{i=1}^{n} l_i} \tag{37}$$

If the leach test option is used, a sample representing the average radionuclide concentration within the zone of contamination should be formed for each soil core by combining discrete samples into a composite sample for the test. The composites should include only samples taken within the zone of contamination (i.e., clean soil below the minimum detectable concentration should not be mixed with contaminated soil).

As with any Superfund sampling effort, all analytical data should be reviewed to ensure that Superfund quality assurance program requirements are met (U.S. EPA, 1993d).

4.2.9 Reporting. The decision process for subsurface soil screening should be thoroughly documented. This documentation should contain as a minimum: a map of the site showing the contaminated soil sources and any areas assumed not to be contaminated, the soil core sampling points within each source, and the soil core sampling points that were compared with the SSLs; the depth and area assumed for each source and their basis; the average soil properties used to calculate SSLs for each source; a description of how samples were taken and (if applicable) how composite samples were formed; the raw analytical data; the average soil core radionuclide concentrations compared with the SSLs for each source; and the results of all QA/QC analyses.

4.3 Basis for the Surface Soil Sampling Strategies: Technical Analyses Performed

This section describes a series of technical analyses conducted to support the sampling strategy for surface soils outlined in the Soil Screening Guidance for chemicals. Section 4.3.1 describes the sample design procedure presented in the December 1994 draft guidance (U.S. EPA, 1994h). The remaining sections describe the technical analyses conducted to develop the final SSL sampling strategy. Section 4.3.2 describes an alternative, nonparametric procedure that EPA considered but **rejected** for the soil screening strategy.

Section 4.3.3 describes the simulations conducted to support the selection of the Max test and the Chen test in the final Soil Screening Guidance for chemicals. These simulation results also can be used to determine sample sizes for site conditions not adequately addressed by the tables in Section 4.1. Quantitation limit and multiple comparison issues are discussed in Sections 4.3.4 and 4.3.5, respectively. Section 4.3.6 describes a limited investigation of compositing samples within individual EA sectors or strata.

4.3.1 1994 Draft Guidance Sampling Strategy. The DQO-based sampling strategy in the 1994 draft Soil Screening Guidance assumed a lognormal distribution for contaminant levels over an EA and derived sample size determinations from lognormal confidence interval procedures by C. E. Land (1971). This section summarizes the rationale for this approach and technical issues raised by peer review.

For the 1994 draft Soil Screening Guidance, EPA based the surface soil SSL methodology on the comparison of the arithmetic mean concentration over an EA with the SSL. As explained in Section 4.1, this approach reflects the type of exposure to soil under a future residential land use scenario. A person moving randomly across a residential lot would be expected to experience an average concentration of contaminants in soil.

Generally speaking, there are few nonparametric approaches to statistical inference about a mean unless a symmetric distribution (e.g., normal) is assumed, in which case the mean and median are identical and inference about the median is the same as inference about the mean. However, environmental contaminant concentration distributions over a surface area tend to be skewed with a long right tail, so symmetry is not plausible. In this case the main options for inference about means are inherently parametric, i.e., they are based on an assumed family of probability distributions.

In addition to being skewed with a long right tail, environmental contaminant concentration data must be positive because concentration measurements cannot be negative. Several standard two-parameter probability models are nonnegative and skewed to the right, including the gamma, lognormal, and Weibull distributions. The properties of these distributions are summarized in Chapter 12 of Gilbert (1987).

The lognormal distribution is the distribution most commonly used for environmental contaminant data (see, e.g., Gilbert, 1987, page 164). The lognormal family can be easy to work with in some respects, due to the work of Land (1971, 1975) on estimating confidence intervals for lognormal parameters, which are also described in Gilbert (1987).

The equation for estimating the Land upper confidence limit (UL) for a lognormal mean has the form

$$UL = \exp\left(\bar{y} + \frac{s_y^2}{2} + \frac{s_y H}{\sqrt{n - 1}} \right) \qquad \textbf{(38)}$$

where y and s_y are the average and standard deviation of the sample log concentrations. The lower confidence limit (LL) has a similar form. The factor H depends on s_y and n and is tabulated in Gilbert (1987) and Land (1975). If the data truly follow a lognormal distribution, then the Land confidence limits are exact (i.e., the coverage probability of a 95 percent confidence interval is 0.95).

The problem formulation used to develop SSL DQOs in the 1994 draft Soil Screening Guidance tested the null hypothesis H_0: $\mu \geq 2$ SSL versus the alternative hypothesis H_1: $\mu < 2$ SSL, with a Type I error rate of 0.05 (at 2 SSL), and a Type II error rate of 0.20 at 0.5 SSL (μ represents the true EA mean). That is, the probability of incorrectly deciding not to investigate further when the true mean is 2 SSL was set not to exceed 0.05, and the probability of incorrectly deciding to investigate further when the true mean is 0.5 SSL was not to exceed 0.20.

This null hypothesis can be tested at the 5 percent level of significance by calculating Land's upper 95 percent confidence limit for a lognormal mean, if one assumes that the true EA concentrations are lognormally distributed. The null hypothesis is rejected if the upper confidence limit falls below 2 SSL.

Simulation studies of the Land procedure were used to obtain sample size estimates that achieve these DQOs for different possible values of the standard deviation of log concentrations. Additional simulation studies were conducted to calculate sample sizes and to investigate the properties of the Land procedure in situations where specimens are composited.

All of these simulation studies assumed a lognormal distribution of site concentrations. If the underlying site distribution is lognormal, then the composites, viewed as physical averages, are not lognormal (although they may be approximately lognormal). Hence, correction factors are necessary to apply the Land procedure with compositing, if the individual specimen concentrations are assumed lognormal. The correction factors were also developed through simulations. The correction factors are multiplied by the sample standard deviation, s_y, before calculating the confidence limit and conducting the test.

Procedures for estimating sample sizes and testing hypotheses about the site mean using the Land procedure, with and without compositing, are described in the 1994 draft Technical Background Document (U.S. EPA, 1994i).

A peer review of the draft Technical Background Document identified several issues of concern:

- The use of a procedure relying strongly on the assumption of a lognormal distribution
- Quantitation limit issues
- Issues associated with multiple hypothesis tests where multiple contaminants are present in site soils.

The first issue is of concern because the small sample sizes appropriate for surface soil screening will not provide sufficient data to validate this assumption. To address this issue, EPA considered several alternative approaches and performed extensive analyses. These analyses are described in Sections 4.3.2 and 4.3.3. Section 4.3.3 describes extensive simulation studies involving a variety of distributions that were done to compare the Land, Chen, and Max tests and to develop the latter two as options for soil screening.

4.3.2 Test of Proportion Exceeding a Threshold. One of the difficulties noted for the Land test, described in Section 4.3.1, is its strong reliance on an assumption of lognormality (see Section 4.3.3). Even in cases where the assumption may hold, there will rarely be sufficient information to test it.

A second criticism of applying the Land test (or another test based on estimating the mean) is that values must be substituted for values reported as less than a quantitation limit (<QL). (As noted in Section 4.3.4, how one does this substitution is of little relevance if the SSL is much larger than the QL. However, even if a moderate proportion of the data values fall below the QL and are censored, then the lognormal distribution may not be a good model for the observed concentrations.)

A third criticism of using the Land test for screening is its requirement for large sample sizes when the contaminant variability across the EA is expected to be large (e.g., a large coefficient of variation). Because of these drawbacks to applying the Land procedure, EPA considered alternative, nonparametric procedures. One such alternative that was considered is the test described below.

For a given contaminant, let P represent the proportion of all possible sampling units across the EA for which the concentration exceeds 2 SSL. In essence, P represents the proportion of the EA with true contaminant levels above 2 SSL. A nonparametric test involving P was developed as follows.

Let P_0 be a fixed proportion of interest chosen in such a way that if that proportion (or more) of the EA has contamination levels above 2 SSL, then that EA should be investigated further. One way to obtain a rough equivalence between the test for a mean greater than 2 SSL and a test involving P is to choose $1-P_0$ to correspond to the percentile of the lognormal distribution at which the mean occurs. One can show that this is equivalent to choosing

$$P_0 = 1 - \Phi[0.5\sigma] = 1 - \Phi[0.5\sqrt{\ln(1+CV^2)}] \quad \textbf{(39)}$$

where

σ = assumed standard deviation of the logarithms of the concentrations
CV = assumed coefficient of variation of the contaminant concentrations
Φ = distribution function of the standard normal distribution.

Here, the fixed proportion P_0 will be less than one-half. The hypotheses are framed as

H_0: $P \geq P_0$ (EA needs further investigation)

versus

H_1: $P < P_0$ (EA does not need further investigation).

The test is based on concentration data from a grid sample of N points in the EA (without compositing). Let p represent the proportion of these n points with observed concentrations greater than or equal to 2 SSL. The test is carried out by choosing a critical value, p_c, to meet the desired Type I error rate, that is,

$$\alpha = \text{Prob}\ (p < p_c \mid P = P_0) = 0.05. \quad \textbf{(40)}$$

The sample size should be chosen to satisfy the Type II error rate at some specified alternative value P_1, where $P_1 < P_0$. For example, to have an 80 percent power at P_1:

$$1-\beta = \text{Prob}\ (p < p_c \mid P = P_1) = 0.80. \quad \textbf{(41)}$$

If the same type of rationale for choosing P_0 (corresponding to 2 SSL) is used to make P_1 correspond to 0.5 SSL, then one would choose

$$P_1 = 1 - \Phi\ [\ 0.5\ \sigma + 1.386/\sigma]. \quad \textbf{(42)}$$

Sample sizes for this test were developed based on the preceding formulation and were found to be approximately the same as those required by the Land procedure, though they tended to be slightly higher than the Land sample sizes for small σ, and slightly smaller for large σ.

The major advantage of this test, in contrast to the Land procedure, for example, is its generality; the only assumption required is that random sampling be used to select the sample points. Its principal disadvantages are:

- Compositing of samples cannot be included (since the calculation of p requires the count of the number of units with observed levels at or above 2 SSL).

- The test does not deal directly with the mean contaminant level at the EA, which is the fundamental parameter for risk calculations.

- Because the test does not depend directly on the magnitude of the concentrations, it is possible that the test will give misleading results relative to a test based on a mean. This can occur, for example, when only a small portion of the EA has very high levels (i.e., a hot spot). In that case, the observed p will converge for increasing n to that proportion of the EA that is contaminated; it would do the same if the concentration levels in that same portion were just slightly above 2 SSL. A test based on a mean for large samples, however, is able to distinguish between these two situations; by its very nature, a test based on a proportion of measurements exceeding a single threshold level cannot.

For these reasons, the test described here based on the proportion of observations exceeding 2 SSL was not selected for inclusion in the current guidance.

4.3.3 Relative Performance of Land, Max, and Chen Tests.

A simulation study was conducted to compare the Land, Chen, and Max tests and to determine sample sizes necessary to achieve DQOs. This section describes the design of the simulation study and summarizes its results. Detailed output from the simulations is presented in Appendix I.

Treatment of Data Below the Quantitation Limit. Review of quantitation limits for 110 chemicals showed that for more than 90 percent of the chemicals, the quantitation limit was less than 1 percent of the ingestion SSL. In such cases, the treatment of values below the QL is not expected to have much effect, as long as all data are used in the analysis, with concentrations assigned to results below the QL in some reasonable way. In the simulations, the QL was assumed to be SSL/100 and any simulated value below the QL was set equal to 0.5 QL. This is a conservative assumption based on the comparison of ingestion SSLs with QLs.

Decision Rules. For the **Land procedure**, as discussed in Section 4.3.1, the null hypothesis H_0: $\mu \geq 2$ SSL (where μ represents the true mean concentration for the EA) can be tested at the 5 percent level by calculating Land's upper 95 percent confidence limit for a lognormal mean. The null hypothesis is rejected (i.e., surface soil contaminant concentrations are less than 2 SSL), if this upper confidence limit falls below 2 SSL. This application of the Land (1971) procedure, as described in the draft 1994 Guidance, will be referred to as the "SSL DQOs" and the "original Land procedure."

For the **Max test**, one decides to walk away if the maximum concentration observed in composite samples taken from the EA does not exceed 2 SSL. As indicated in Section 4.1.6, it is viewed as providing a test of the original null hypothesis, H_0: $\mu \geq 2$ SSL. The Max test does not inherently control either type of error rate (i.e., its critical region is always the region below 2 SSL, not where concentrations below a threshold that achieve a specified Type I error rate). However, control of error rates for the Max test can be achieved through the DQO process by choice of design (i.e., by choice of the number N of composite samples and choice of the number C of specimens per composite).

The **Chen test** requires that the null hypothesis have the form H_0: $\mu \leq \mu_0$, with the alternative hypothesis as H_1: $\mu > \mu_0$ (Chen, 1995). Hypotheses or DQOs of this form are referred to as "flipped hypotheses" or "flipped DQOs" because they represent the inverse of the actual hypothesis for SSL decisions. In the simulations, the Chen method was applied with $\mu_0 = 0.5$ SSL at significance levels (Type I error rates) of 0.4, 0.3, 0.2, 0.1, 0.05, 0.025, and 0.01. In this formulation, a Type I error occurs if one decides incorrectly to investigate further when the true site mean, μ, is at or below 0.5 SSL.

The two formulations of the hypotheses are equivalent in the sense that both allow achievement of soil screening DQOs. That is, working with either formulation, it is possible to control the probability of incorrectly deciding to walk away when the true site mean is 2 SSL and to also control the probability of incorrectly deciding to investigate further when the true site mean is 0.5 SSL.

In addition to the original Land procedure, the Chen test, and the Max test, the simulations also include the Land test of the flipped null hypothesis H_0: $\mu \leq 0.5$ SSL at the 10 percent significance level. This Land test of the flipped hypothesis was included to investigate how interchanging the null and alternative hypotheses affected sample sizes for the Land and Chen procedures.

Simulation Distributions. In the following description of the simulations, parameter acronyms used as labels in the tables of results are indicated by capital letters enclosed in parentheses.

Each distribution used for simulation is a mixture of a lower concentration distribution and a higher concentration distribution. The lower distribution represents the EA in its natural (unpolluted) state, and the higher distribution represents contaminated areas. Typically, all measurements of pollutants in uncontaminated areas are below the QL. Accordingly, the lower distribution is assumed to be completely below the QL. For the purposes of this analysis, it is unnecessary to specify any other aspect of the lower distribution, because any measurement below the QL is set equal to 0.5 QL.

A parameter between 0 and 1, called the mixing proportion (MIX), specifies the probability allocated to the lower distribution. The remaining probability (1-MIX) is spread over higher values according to either a lognormal, gamma, or Weibull distribution. The parameters of the higher distribution are chosen so that the overall mixture has a given true EA mean (MU) and a given coefficient of variation (CV). Where s is the sample standard deviation, $\bar{x}$ is the sample mean, and C is the number of specimens per composite sample, CV is defined as:

$$CV = \frac{s}{\bar{x}} \quad or \quad CV = \frac{\left(\sqrt{C}\right)s}{\bar{x}} \tag{43}$$

The following parameter values were used in the simulations:

EA mean (MU) = 0.5 SSL or 2 SSL

EA coefficient of variation (CV) = 1, 1.5, 2, 2.5, 3, 3.5, 4, 5, or 6 (i.e., 100 to 600 percent)

Number of specimens per composite (C) = 1, 2, 3, 4, 5, 6, 8, 9, 12, or 16

Number of composites chemically analyzed (N) = 4, 5, 6, 7, 8, 9, 12, or 16.

The true EA mean was set equal to 0.5 SSL or 2 SSL in order to estimate the two error rates of primary concern. Most CVs encountered in practice probably will lie between 1 and 2.5 (i.e., variability between 100 and 250 percent). This expectation is based on data from the Hanford site (see Hardin and Gilbert, 1993) and the Piazza Road site (discussed in Section 4.3.6). EPA believes that the most practical choices for the number of specimens per composite will be four and six. In some cases, compositing may not be appropriate (the case C = 1 corresponds to no compositing). EPA also believes that for soil screening, a practical number of samples chemically analyzed per EA lies below nine, and that screening decisions about soils in each EA should not be based on fewer than four chemical analyses.

For a given CV, there is a theoretical limit to how large the mixing proportion can be. The values of the mixing proportion used in the simulations are shown below as a function of CV. The case MIX = 0 corresponds to an EA characterized by a gamma, lognormal, or Weibull distribution. A value of MIX near 1 indicates an EA where all concentrations are below the QL except those in a small portion of the EA. Neither of these extremes implies an extreme overall mean. If MIX = 0, the contaminating (higher) distribution can have a low mean, resulting in a low overall mean. If MIX is near 1 (i.e., a relatively small contamination area), a high overall mean can be obtained if the mean of the distribution of contaminant concentrations is high enough.

CV	Values of MIX used in the simulations
1.0	0, 0.49
1.5	0, 0.50
2.0	0, 0.50, 0.75
2.5	0, 0.50, 0.85
3.0	0, 0.50, 0.85
3.5	0, 0.50, 0.90
4.0	0, 0.50, 0.90
5.0	0, 0.50, 0.95
6.0	0, 0.50, 0.95

Treatment of Measurement Error. Measurement errors were assumed to be normally distributed with mean 0 (i.e., unbiased measurements) and standard deviation equal to 20 percent of the true value for each chemically analyzed sample. (Earlier simulations included measurement error standard deviations of 10 percent and 25 percent. The difference in results between these two cases was negligible.)

Number of Simulated Samples. Unique combinations of the simulation parameters considered (i.e., 2 values of the EA mean, 10 values for the number of specimens per composite, 8 values for the number of composite samples, 25 combinations of CV and MIX, and 3 contamination models—lognormal, gamma, Weibull), result in a total of 12,000 simulation conditions. One thousand simulated random samples were generated for each of the 12,000 cases obtained by varying the simulation parameters as described above. The average number of physical samples simulated from an EA for a hypothesis test (i.e., the product CN) was 56.

The following 10 hypothesis tests were applied to each of the 12 million random samples:

- Chen test at significance levels of 0.4, 0.3, 0.2, 0.1, 0.05, 0.025, and 0.01
- Original Land test of the null hypothesis H_0: $\mu \geq 2$ SSL at the 5 percent significance level
- Land test of the flipped null hypothesis H_0: $\mu \leq 0.5$ SSL at the 10 percent significance level
- Maximum test.

These simulations involved generation of approximately 650 million random numbers.

Simulation Results. A complete listing of the simulation results, with 150 columns and 59 lines per page, requires 180 pages and is available from EPA on a 3.5-inch diskette.

Representative results for gamma contamination data, with eight composite samples that each consist of six specimens, are shown in Table 4.9. The gamma contamination model is recommended for determining sample size requirements because it was consistently seen to be least favorable, in the sense that it required higher sample sizes to achieve DQOs than either of the lognormal or Weibull models. Hence, sample sizes sufficient to protect against a gamma distribution of contaminant concentrations are also protective against a lognormal or Weibull distribution.

Table 4.9 Comparison of Error Rates for Max Test, Chen Test (at .20 and .10 Significance Levels), and Original Land Test, Using 8 Composites of 6 Samples Each, for Gamma Contamination Data

MU/SSL	MIX	Max test	0.20 Chen test	0.10 Chen test	Land test
C=6 N=8 CV=4					
0.5	.00	.35	.18	.09	.99
0.5	.50	.40	.22	.11	.99
0.5	.90	.40	.19	.09	.98
2.0	.00	.06	.10	.18	.00
2.0	.50	.06	.11	.18	.00
2.0	.90	.04	.16	.29	.01
C=6 N=8 CV=3					
0.5	.00	.24	.18	.10	.93
0.5	.50	.25	.19	.10	.94
0.5	.85	.23	.22	.11	.99
2.0	.00	.04	.03	.06	.00
2.0	.50	.03	.03	.05	.00
2.0	.85	.03	.06	.12	.00
C=6 N=8 CV=2					
0.5	.00	.07	.22	.11	.57
0.5	.50	.06	.19	.09	.68
0.5	.75	.04	.19	.10	.85
2.0	.00	.02	.00	.00	.01
2.0	.50	.02	.00	.01	.00
2.0	.75	.01	.00	.01	.00
C=6 N=8 CV=1					
0.5	.00	.00	.20	.10	.01
0.5	.49	.00	.20	.12	.12
2.0	.00	.01	.00	.00	.02
2.0	.49	.01	.00	.00	.00

MU = True EA Mean - see subsection entitled ?Simulation Distributions” in Section 5.3 .3.
MIX = Mixing Proportion - see subsection entitled ?Simulation Distributions” in Section 5.3 .3
C = Number of specimens in a composite.
N = Number of composites analyzed.
CV = EA coefficient of variation $\frac{\left(\sqrt{C}\right)s}{\bar{x}}$

where s = sample standard deviation and $\bar{x}$ = mean sample concentration

Table 4.9 shows that the original Land method is unable to control the error rates at 0.5 SSL for gamma distributions. This limitation of the Land method was seen consistently throughout the results for all nonlognormal distributions tested. This limitation led to removal of the Land procedure from the Soil Screening Guidance.

Earlier simulation results for gamma and Weibull distributions did not censor results below the QL and used pure unmixed distributions. In these cases, as the sample size N increased, with all other factors fixed, the Land error rates at 0.5 SSL increased toward 1. Normally, the expectation is that as the sample size increases, information increases, and error rates decrease.

When using data from a Weibull or gamma distribution, the Land confidence interval endpoints converge to a value **that does not equal** the true site mean, μ_x , and results in an increase in error rates. This phenomenon is easily demonstrated, as follows. Let X denote the concentration random variable, let Y = ln(X) denote its logarithm. Let μ_y and σ_y denote the mean and standard deviation of logarithms of the soil concentrations. Then, as the sample size increases, the Land confidence interval endpoints (UL and LL) converge to

$$UL = LL = \exp(\mu_y + \frac{\sigma_y^2}{2}) \quad \textbf{(44)}$$

If X is lognormally distributed, this expression is the mean of X. If X has a Weibull or gamma distribution, this expression is **not** the mean of X. This inconsistency accounts for the increase in error rates with sample size.

Table 4.9 also shows the fundamental difference between the Max test and the Chen test. For the Max test, the probability of error in deciding to walk away when the EA mean is 2.0 SSL is fairly stable, ranging from 0.01 to 0.06 across the different values of the CV. On the other hand, these error rates vary more across the CV values for the Chen test (e.g., from 0.00 to 0.29 for Chen test at the 0.10 significance level). This occurs because the Chen test is designed to control the other type of error rate (at 0.5 SSL). The Max test is presented in the 1995 Soil Screening Guidance (U.S. EPA, 1995c) because of its simplicity and the stability of its control over the error rate at 2 SSL.

Table 4.10 shows error rate estimates for four to nine composite samples that each consist of four, six, or eight specimens for EAs with CVs of 2, 2.5, 3, or 3.5, and assuming a gamma distribution. Table 4.10 should be adequate for most SSL planning purposes. However, more complete simulation results are reported in Appendix I.

Planning for CVs at least as large as 2 is recommended because it is known that CVs greater than 2 occur in practice (e.g., for two of seven EAs in the Piazza Road simulations reported in Section 4.3.6). One conclusion that can be drawn from Table 4.11 is that composite sample sizes of four are often inadequate. Further support for this conclusion is reported in the Piazza Road simulations discussed in Section 4.3.6.

Conclusions. The primary conclusions from the simulations are:

- For distributions other than lognormal, the Land procedure is prone to decide to investigate further at 0.5 SSL, when the correct decision is to walk away. It is therefore unsuitable for surface soil screening.
- Both the Max test and the Chen test perform acceptably under a variety of distributional assumptions and are potentially suitable for surface soil screening.

Table 4.10. Error Rates of Max Test and Chen Test at .2 (C20) and .1 (C10) Significance Level for CV = 2, 2.5, 3, 3.5

		CV = 2.0			CV = 2.5			CV = 3.0			CV = 3.5		
N	**MU/SSL**	**Max**	**C20**	**C10**	**Max**	**C20**	**C10**	**Max**	**C20**	**C10**	**Max**	**C20**	**C10**
C = 4													
4	0.5	.09	.20	.11	.14	.18	.09	.19	.18	.08	.24	.20	.10
4	2.0	.13	.08	.16	.19	.17	.28	.20	.21	.33	.26	.29	.42
5	0.5	.11	.21	.10	.15	.18	.09	.26	.20	.08	.26	.20	.09
5	2.0	.10	.05	.11	.10	.09	.18	.17	.19	.30	.18	.23	.36
6	0.5	.11	.21	.12	.21	.20	.10	.28	.21	.11	.31	.19	.09
6	2.0	.06	.03	.08	.08	.08	.14	.11	.13	.23	.11	.18	.28
7	0.5	.12	.20	.10	.25	.22	.11	.31	.20	.09	.36	.18	.10
7	2.0	.04	.03	.05	.05	.04	.09	.08	.11	.18	.08	.14	.23
8	0.5	.16	.19	.09	.25	.20	.09	.36	.20	.10	.42	.20	.09
8	2.0	.02	.02	.03	.04	.03	.07	.05	.08	.14	.07	.13	.21
9	0.5	.16	.21	.11	.28	.20	.09	.36	.18	.09	.44	.22	.12
9	2.0	.01	.01	.02	.03	.03	.06	.04	.07	.13	.07	.12	.20
C = 6													
4	0.5	.03	.20	.12	.08	.21	.12	.15	.20	.10	.16	.17	.08
4	2.0	.14	.03	.08	.16	.08	.17	.17	.14	.24	.20	.19	.33
5	0.5	.04	.20	.10	.11	.17	.09	.17	.20	.10	.22	.20	.10
5	2.0	.09	.02	.05	.09	.04	.10	.13	.10	.18	.15	.13	.24
6	0.5	.06	.20	.11	.14	.21	.10	.19	.20	.10	.25	.20	.10
6	2.0	.04	.01	.02	.06	.03	.07	.09	.07	.14	.09	.10	.19
7	0.5	.06	.20	.09	.12	.19	.10	.23	.22	.10	.29	.21	.10
7	2.0	.02	.00	.01	.05	.02	.04	.06	.06	.10	.08	.09	.14
8	0.5	.06	.19	.09	.15	.20	.10	.25	.19	.10	.30	.19	.10
8	2.0	.02	.00	.01	.02	.01	.03	.03	.03	.05	.04	.06	.11
9	0.5	.06	.20	.10	.18	.22	.11	.28	.20	.11	.34	.19	.09
9	2.0	.01	.00	.01	.02	.01	.02	.03	.02	.04	.03	.05	.09
C = 8													
4	0.5	.02	.21	.13	.06	.19	.10	.10	.21	.10	.14	.18	.08
4	2.0	.12	.02	.05	.15	.04	.09	.17	.09	.17	.19	.14	.25
5	0.5	.03	.22	.11	.05	.20	.11	.11	.20	.10	.17	.19	.09
5	2.0	.07	.01	.02	.09	.02	.06	.09	.04	.10	.12	.08	.17
6	0.5	.02	.18	.09	.08	.21	.11	.13	.19	.10	.20	.20	.10
6	2.0	.04	.00	.01	.06	.01	.02	.07	.04	.07	.08	.07	.13
7	0.5	.03	.20	.11	.09	.20	.11	.18	.21	.11	.22	.20	.11
7	2.0	.03	.00	.00	.04	.01	.01	.04	.02	.04	.05	.05	.09
8	0.5	.04	.20	.10	.11	.21	.11	.17	.21	.10	.26	.19	.10
8	2.0	.02	.00	.00	.02	.01	.01	.04	.01	.03	.03	.03	.06
9	0.5	.04	.21	.11	.11	.21	.10	.20	.19	.10	.30	.23	.12
9	2.0	.01	.00	.00	.02	.00	.01	.01	.00	.01	.02	.02	.04

MU = True EA Mean - see subsection entitled ?Simulation Distributions” in Section 5.3 .3.
MIX = Mixing Proportion - see subsection entitled ?Simulation Distributions” in Section 5.3 .3
C = Number of specimens in a composite.
N = Number of composites analyzed.
CV = EA coefficient of variation $\frac{\left(\sqrt{C}\right)s}{\bar{x}}$

where s = sample standard deviation and $\bar{x}$ = mean sample concentration

4.3.4 Treatment of Observations Below the Limit of Quantitation. Test procedures that are based on estimating a mean contaminant level for an EA, such as the Land and Chen procedures, make use of each measured concentration value. For this reason, the use of all reported concentration measurements in such calculations should be considered regardless of their magnitude—that is, even if the measured levels fall below a quantitation level. One argument for this approach is that the QL is itself an estimate. Another is that some value will have to be substituted for any censored data point (i.e., a point reported as <QL), and the actual measured value is at least as accurate as a substituted value.

The peer review of the Draft Soil Screening Guidance raised the following issue:

> If such censored values do occur in a data set, what values should be used?

There is a substantial amount of literature on this subject and a variety of sophisticated approaches. In the context of SSLs, however, a simple approach is recommended. Consistent with general Superfund guidance, each observation reported as "<QL" shall be replaced with 0.5 QL for computation of the sample mean.

The evidence suggests that the ingestion SSL generally will be 2 orders of magnitude or more greater than the QL for most contaminants. In these cases, the results of soil screening will be insensitive to alternative procedures that could be used to substitute values for observations reported as "<QL." When the SSL is not much greater than the QL (e.g., SSL < 50 QL), the outcome of the soil screening could be affected by the procedure used to substitute for "<QL" values.

The most conservative approach would be to substitute the concentration represented by the QL itself for all observations reported as "<QL." In the context of the SSLs, however, the simple approach of using 0.5 QL is suggested. This will be sufficiently conservative given the conservative factors underlying the SSLs.

4.3.5 Multiple Hypothesis Testing Considerations. The Soil Screening Guidance addresses the following hypothesis testing problem for each EA:

> H_0: mean concentration of a given chemical $\geq$ 2 SSL

versus

> H_1: mean concentration of a given chemical < 2 SSL.

The default value for the probability of a Type I error is $\alpha = 0.05$, while the default value for the power of the test at 0.5 SSL is $1-\beta = 0.80$. The test is applied separately for each chemical, so that these probabilities apply for each individual chemical. Thus, there is an 80 percent probability of walking away from an EA (i.e., rejecting H_0) when only one chemical is being tested and its true mean level is 0.5 SSL and a 5 percent probability of walking away if its true mean level is 2 SSL.

However, the Soil Screening Guidance does not explicitly address the following issues:

> What is the composite probability of walking away from an EA if there are multiple contaminants?

and

> If such probabilities are unacceptable, how should one compensate when testing for multiple contaminants within a single EA?

The answer to the first question cannot be determined, in general, since the concentrations of the various contaminants will often be dependent on one another (e.g., this would be expected if they originated from the same source of contamination). The joint probability of walking away can be determined, however, if one makes the simplifying assumption that the contaminant concentrations for the different chemicals are independent (uncorrelated). In that case, the probability of walking away is simply the product of the individual rejection probabilities.

For two chemicals (Chemical A and Chemical B, say), this is:

Pr{walking away from EA} = Pr{reject H_0 for Chemical A} x Pr{reject H_0 for Chemical B}.

While these joint probabilities must be regarded as approximate, they nevertheless serve to illustrate the effect on the error rates when dealing with multiple contaminants.

Assume (for illustrative purposes only) that the probabilities for rejecting the null hypothesis (walking away from the EA) for each single chemical appear as follows:

True concentration	**Probability of rejecting H_0**
0.2 SSL	0.95
0.5 SSL	0.80 (default 1-β)
0.7 SSL	0.60
1.0 SSL	0.50
1.5 SSL	0.20
2.0 SSL	0.05 (default α)

Let C(A) denote the concentration of Chemical A divided by the SSL, and let P(A) denote the corresponding probability of rejecting H_0. Define C(B) and P(B) similarly for Chemical B. Assuming independence, the joint probabilities of rejecting the null hypothesis (walking away) are as shown in Table 45.11.

Table 4.11. Probability of "Walking Away" from an EA When Comparing Two Chemicals to SSLs

Chemical A		**Chemical B**					
C(A)	**P(A)**	**C(B) = 0.2 P(B) =.95**	**C(B) = 0.5 P(B) = .80**	**C(B) = 0.7 P(B) = .60**	**C(B) = 1.0 P(B) = .50**	**C(B) = 1.5 P(B) = .20**	**C(B) = 2.0 P(B) = .05**
0.2	0.95	0.90	0.76	0.57	0.48	0.19	0.05
0.5	0.80	0.76	0.64	0.48	0.40	0.16	0.04
0.7	0.60	0.57	0.48	0.36	0.30	0.12	0.03
1.0	0.50	0.48	0.40	0.30	0.25	0.10	0.03
1.5	0.20	0.19	0.16	0.12	0.10	0.04	0.01
2.0	0.05	0.05	0.04	0.03	0.03	0.01	<0.01

These probabilities demonstrate that the test procedure will tend to be very conservative if multiple chemicals are involved—that is, **all** of the chemical concentrations must be quite low relative to their SSL in order to have a high probability of walking away from the EA. On the other hand, there will be a high probability that further investigation will be called for if the mean concentration for even a single chemical is twice the SSL.

A potential problem occurs when there are several chemicals under consideration and when all or most of them have levels slightly below the SSL (e.g., near 0.5 SSL). For instance, if each of six independent chemicals had levels at 0.5 SSL, the probability of rejecting the null hypothesis would be 80 percent for each such chemical, but the probability of walking away from the EA would be only $(0.80)^6 = 0.26$.

If the same samples are being analyzed for multiple chemicals, then the original choice for the number of such samples ideally should have been based on the worst case (i.e., the chemical expected to have the largest variability). In this case, the probability of correctly rejecting the null hypothesis at 0.5 SSL for the chemicals with less variability will be higher. The overall probability of walking away will be greater than shown above if all or some of the chemicals have less variability than assumed as the basis for determining sample sizes. Here, the sample size will be large enough for the probability of rejecting the null hypothesis at 0.5 SSL to be greater than 0.80 for these chemicals.

The probability values assumed above for deciding that no further investigation is necessary for individual chemicals, which are the basis for these conclusions, are equally applicable for the Land, Chen, and Max tests. They simply represent six hypothetical points of the power curves for these tests (from 0.2 SSL to 2.0 SSL). Therefore, the conclusions are equally applicable for each of the hypothesis testing procedures that have been considered in the current guidance for screening surface soils.

If the surface soil concentrations are positively correlated, as expected when dealing with multiple chemicals, then it is likely that either all the chemicals of concern have relatively high concentrations or they all have relatively low concentrations. In this case, the probability of making the correct decision for an EA would be greater than that suggested by the above calculations that assume independence of the various chemicals.

However, the potential problem of several chemicals having concentrations near 0.5 SSL is not precluded by assuming positive correlations. In fact, it suggests that if the EA average for one chemical is near 0.5 SSL, then the average for others is also likely to be near 0.5 SSL, which is exactly the situation where the probability of **not** walking away from the EA can become large because there is a high probability that H_0 will be rejected for at least one of these chemicals.

An alternative would be to use multiple hypothesis testing procedures to control the overall error rate for the set of chemicals (i.e., the set of hypothesis tests) rather than the separate error rates for the individual chemicals. Guidance for performing multiple hypothesis tests is beyond the scope of the current document. Obtain the advice of a statistician familiar with multiple hypothesis testing procedures if the overall error rates for multiple chemicals is of concern for a particular site. The classical statistical guidance regarding this subject is *Simultaneous Statistical Inference* (Miller, 1991).

4.3.6 Investigation of Compositing Within EA Sectors. If one decides that an EA needs further investigation, then it is natural to inquire which portion(s) of the EA exceed the screening level. This is a different question than simply asking whether or not the EA average soil concentration exceeds the SSL. Conceivably, this question may require additional sampling, chemical analysis, and statistical analysis. A natural question is whether this additional effort can be avoided by forming composites within sectors (subareas) of the EA. The sector with the highest estimated concentration would then be a natural place to begin a detailed investigation.

The simulations to investigate the performance of rules to decide whether further investigation is required, reported in Section 4.3.3, make specific assumptions about the sampling design. It is assumed that N composite samples are chemically analyzed, each consisting of C specimens selected to be statistically representative of the entire EA. The key point, in addition to random sampling, is that composites must be formed **across** sectors rather than **within** sectors. This assumption is necessary to achieve composite samples that are representative of the EA mean (i.e., have the EA mean as their expected value).

If compositing is limited to sectors, such as quadrants, then each composite represents its sector, rather than the entire EA. The simulations reported in Section 4.3.3, and sample sizes based on them, do not apply to this type of compositing. This does not necessarily preclude compositing within sectors for both purposes, i.e., to test the hypothesis about the EA mean and also to indicate the most contaminated sector. However, little is known about the statistical properties of this approach when applying the Max test, which would depend on specifics of the actual spatial distribution of contaminants for a given EA. Because of the lack of extensive spatial data sets for contaminated soil, there is limited basis for determining what sample sizes would be adequate for achieving desired DQOs for various sites. However, one spatial data set was available and used to investigate the performance of compositing within sectors at one site.

Piazza Road Simulations. Data from the Piazza Road NPL site were used to investigate the properties of tests of the EA mean based on compositing within sectors, as compared to compositing between sectors. The investigation of a single site cannot be used to validate a given procedure, but it may indicate whether further investigation of the procedure is worthwhile.

Seven nonoverlapping 0.4-acre EAs were defined within the Piazza Road site. Each EA is an 8-by-12 grid composed of 14'x14' squares. The data consist of a single dioxin measurement of a composite sample from each small square. These measurements are regarded as true values for the simulations reported in this section. Measurement error was incorporated in the same fashion as for the simulations reported in Section 4.3.3.

Each of the seven EAs was subdivided into four 4-by-6 sectors, six 4-by-4 sectors, eight 4-by-3 sectors, twelve 2-by-4 sectors, and sixteen 2-by-3 sectors. Results are presented here for the cases of four, six, and eight sectors because composites of more than eight specimens are expected to be used rarely, if at all.

Table 4.12 presents the "true" mean and CV for each EA, computed from all 96 measurements within the 0.4-acre EA. The CVs range from 1.0 to 2.2. Note that two of the seven CVs equal or exceed 2 at this site. This supports EPA's belief that at many sites it is prudent, when planning sample size requirements for screening, to assume a CV of at least 2.5 and to consider the possibility of CVs as large as 3 or 3.5.

As data on variability within EAs for different sites and contaminant conditions accrue over time, it will be possible to base the choice of procedures on a larger, more comprehensive database, rather than just a single site.

Appendix J contains results of simulations from the seven Piazza Road EAs. Sampling with replacement from each sector was used, because this was felt to be more consistent with the planned compositing. To estimate the error rates at 0.5 SSL and 2 SSL for each EA, the SSL was defined so that the site mean first was regarded as 0.5 SSL and then was regarded as 2 SSL.

Table 4.12. Means and CVs for Dioxin Concentrations for 7 Piazza Road Exposure Areas

EA	Mean of EA	CV of EA	N
1	2.1	1.0	96
2	2.4	1.6	96
3	5.1	1.1	96
4	4.0	1.2	96
5	9.3	2.0	96
6	15.8	2.2	96
7	2.8	1.4	96

Notation for Results from Piazza Road Simulations. The following notation is used in Appendix J. The design variable (DES) indicates whether compositing was within sector (DES=W) or across sectors (DES=X). As in Section 4.3.3, C denotes the number of specimens per composite, and N denotes the number of composite samples chemically analyzed. Results in Appendix J are for the Chen test at the 10 percent significance level and for the Max test. The true mean and CV are shown in the header for each EA.

Results and Conclusions from Piazza Road Simulations. Although the results from a single site cannot be assumed to apply to all sites, the following observations can be made based on the Piazza Road simulations reported in Appendix J.

- The error rate at 0.5 SSL for the Chen test, using compositing across sectors (DES=X), is generally close to the nominal rate of 0.10. For compositing within sectors (DES=W), the error rate for Chen at 0.5 SSL is generally much lower than the nominal rate.

- Except for plans involving only four analyses (N = 4), the error rate at 2 SSL is always below 0.05 for the Chen test. For the Max test, the error rate at 2 SSL fluctuated between 0 and 16 percent. The error rate at 2 SSL is smaller for the Chen test at the 10 percent significance level than for the Max test in virtually all cases. The only two exceptions to this are for compositing within sector (DES=W) in EA No. 6.

- This observation provides further support for the conclusion drawn from the simulations reported in Section 4.3.3: plans involving only four analyses can result in high error rates in determining the mean contaminant concentration of an EA with the Max test. In most cases the error rates of concern to EPA (at 2 SSL) are 0.10 or larger.

- In general, error rates estimated from Piazza Road simulations for compositing across sectors are at least as small as would be predicted on the basis of the simulation results reported in Section 4.3.3.

- The simulation results show that compositing within sectors using the Max test may be an option for site managers who want to know whether one sector of an EA is more contaminated than the other. However, use of the Max test when compositing within sectors may lead the site manager to draw conclusions about the mean contaminant concentration in that sector only, not across the entire EA.

Part 5: RADIONUCLIDE-SPECIFIC PARAMETERS

Radionuclide-specific parameters required for calculating soil screening levels include an area correction factor (ACF) for the external radiation exposure pathway, and the soil-water partition coefficient for radionuclides (K_d). This part of the background document describes the collection and compilation of these parameters for the SSL radionuclides.

5.1 Area Correction Factor - ACF (unitless)

The risk model used for the external radiation exposure pathway in this guidance effectively assumes that an individual is exposed to a source geometry that is effectively an infinite slab. The concept of an "infinite slab" means that the thickness of the contaminated zone and its aerial extent are so large that it behaves as if it were infinite in its physical dimensions. In practice, soil contaminated to a depth greater than about 15 cm and with an aerial extent greater than about 1,000 m^2 (i.e., one-quarter acre) will create a radiation field approaching that of an infinite slab.

This infinite slab assumption has been used in the calculation of radionuclide slope factors presented in Section 2.1. For very small areas of contamination, this will result in overly conservative estimates of risk. For calculation of SSLs for a residential setting, an adjustment for source area is considered to be an important modification for Superfund sites. Thus, an area correction factor, ACF, has been added to the SSL calculation.

Table 5.1 provides recommended ACFs for radionuclides as a function of source area calculated using MicroShield V5.01.[1] Since the default source size is 0.5-acre (i.e., 2,000 m^2), the default ACF for SSL equations is set at 0.9. The calculations assume for a uniform layer of contamination 15 cm deep with a soil density of 1.6 g/cm^3. A single recommended value is considered suitable for all radionuclides over the range of source areas since EPA's analysis shows that ACFs vary little from one radionuclide to another. For other source areas, recommended ACFs are presented in Table 5.1.

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[1] Grove Engineering, Rockville, MD.

Table 5.1 Recommended Area Correction Factors as Function of Source Area

Source Area (m^2)	ACF
10,000	1.00
5,000	0.94
2,000	0.90
1,000	0.88
500	0.86
100	0.75
50	0.66
10	0.40

EPA's analysis of ACFs is found in Table 5.2, which provides examples of ACFs for several radionuclides as a function of source area calculated using MicroShield V5.01. The calculations assume for a uniform layer of contamination 15 cm deep with a soil density of 1.6 g/cm^3. Strong gamma-ray emitters like ^{60}Co have relatively large slope factors for this pathway relative to the slope factors for weak photon emitters like ^{239}Pu. However, as noted in the table, ACFs vary little from one radionuclide to another over the range of source areas shown. Users that have one of the radionuclides in Table 5.2 as a contaminant at their site may use the radionuclide specific ACF that is appropriate for their source area rather than the value found in Table 5.1.

Table 5.2 Area Correction Factors as Function of Source Area for Selected Radionuclides Calculated Using MicroShield

Source Area (m^2)	^{231}Am	^{60}Co	^{137}Cs	^{239}Pu	^{226}Ra+D	^{232}Th	^{238}U+D
10,000	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5,000	0.93	0.95	0.95	0.94	0.95	0.94	0.94
2,000	0.89	0.92	0.92	0.90	0.92	0.89	0.91
1,000	0.87	0.90	0.90	0.89	0.90	0.88	0.89
500	0.85	0.87	0.87	0.86	0.87	0.86	0.86
100	0.76	0.75	0.76	0.78	0.75	0.77	0.76
50	0.69	0.66	0.67	0.71	0.66	0.70	0.68
10	0.44	0.38	0.39	0.45	0.38	0.44	0.41

5.2 Soil-Water Distribution Coefficients (Kd) for Radionuclides

As with organic chemicals, development of SSLs for inorganics (including radionuclides) requires a soil-water partition coefficient or distribution cofficient (K_d) for each constituent. The soil- water distribution coefficient is typically defined in fate and contaminant transport calculation as the ratio of the contaminant concentrations associated with the solid to the contaminant concentration in the surrounding aqueous solution when the system is at equilibrium (EPA 1999b). However, the simple relationship between soil organic carbon content and sorption observed for organic chemicals does not apply to inorganics (including radionuclides). The soil-water distribution coefficient (K_d) for inorganics (including radionuclides) is affected by numerous geochemical parameters and processes, including pH; sorption to clays, organic matter, iron oxides, and other soil constituents; oxidation/reduction conditions; major ion chemistry; and the chemical form of the radionuclide. The number of significant influencing parameters, their variability in the field, and differences in experimental methods result in as much as seven orders of magnitude variability in measured metal K_d values reported in the literature (see Table 43 in the Soil Screening Guidance: Technical Background Document (EPA 1996b)). This variability makes it much more difficult to derive generic K_d values for metals (including radionuclides) than for organics. Therefore, it is recommended that K_d values be measured for site-specific conditions. If the K_d is not measured site-specifically, then a conservative Kd should be used in calculating SSLs.

Tables 5.3 in the TBD (also C.2a in the User Guide) and C.2b in the User Guide list the default K_d values for each element. Table 5.2 is derived from the EPA Office of Radiation and Indoor Air's 1999 final document *Understanding Variation In Partition Coefficient, K_d, Values, Volume 1: The K_d Model of Measurement, and Application of Chemical Reaction Codes, & Volume 2: Review of Geochemistry and Available K_d Values for Cadmium, Cesium, Chromium, Lead , Plutonium, Radon, Strontium, Thorium, Tritium, and Uranium*. This document is intended to provide technical information to EPA and/or DOE transport modelers on the key geochemical processes affecting contaminant transport through soil and sediments. Particular attention is directed at providing an understanding of: 1) the use of distribution coefficients, 2) the difference between the original thermodynamic K_d parameter derived from the ion-exchange literature and its "empiricized" use in contaminant transport codes, 3) the explicit and implicit assumptions underlying the use of the K_d paramter in contaminant transport codes, and 4) methods for site-specific measurements.

When estimating migration of contaminants from soil to groundwater for a contaminant which is not represented with a default K_d value in either Table 5.3 or Tables C.2a and C.2b in the User Guide, site decision-makers should measure a site-specific K_d. Site decision-makers also may develop site-specific K_ds to more accurately estimate contaminant migration rather than using the default values in Tables 5.4 to 5.9 or either Tables C.2a or C.2b or in the User Guide.

The elements chosen for study in EPA (1999b) include the following: chromium, cadmium, cesium, lead, plutonium, radon, strontium, thorium, tritium, and uranium. The selection of these contaminants by EPA staff was based on two criteria. First, the elements had to be one of high priority to the site remediation or risk assessment activities of EPA and/or DOE (EPA 1993). Second, six nonexclusive categories were developed based on chemical behavior:

1) Cations - cadmium, cesium, plutonium, strontium, thorium, and uranium;
2) Anions - chromium (as CrO_4^{2-});
3) Radionuclides - cesium, plutonium, radon, strontium, thorium, tritium, and uranium;
4) Conservatively Transported Contaminants - tritium and radon;
5) Nonconservatively Transported Contaminants - other than tritium and radon; and
6) Redox Sensitive Elements - chromium, plutonium, and uranium.

By categorizing the contaminants in this manner, general geochemical behaviors of one contaminant may be extrapolated by analogy to other contaminants in the same category. For example, contaminants present primarily in anionic form, such as Cr(VI), tend to adsorb to a limited extent to geological materials (soil, rocks, and geological deposits, hence forth simply referred to as soils). Thus, one might generalize that other anions, such as nitrate, chloride, and U(VI)-anionic complexes, would also adsorb to a limited extent. Literature on the absorption of these three solutes shows no or very little adsorption. Volume 2, EPA (1999) identifies, when possible, minimum and maximum conservative K_d values for each contaminants as a function of key geochemical processes affecting sorption. The tables that follows reflect this conservatism in using the lower bounding K_d values for these contaminants. Site specific measurements of K_d values should be used in determining SSLs. If K_d is not measured site-specifically, then conservative K_d values should be used from the following tables when calculating SSLs.

Table 5.3 Default K_d Values for Selected Elements, based on EPA (1999b).

Element	K_d value* (ml/g)	Element	K_d value (ml/g)
Cs	10	Sr	1
H	0	Th	20
Pu	5	U	0.4
Rn	0		

*Note: K_d values were given of units in ml/g inEPA (1999b). The units in other parts of this TBD are given in L/kg. However, these are equivalent units, that is: 1 ml/g = 1 L/kg

The K_d values in Table 5.3 reflect the most conservative values provided for each element in EPA (1999). Each of these values are based on a chemical mechanism that was considered to provide the most conservative K_d value for that element. Users that have measured pH values at their site that differ from the range given in this report, may want to consult Tables 5.4 to 5.9 for alternative K_ds that are still conservative.

Table 5.4 shows the estimated conservative K_d values for cesium based on cation exchange capacity (CEC) (in meg/100 g) or clay content (in wt.%). This table is for systems containing low concentrations of mica-like minerals (i.e., <5% of the clay-size fraction). This table pertains to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (<0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.

Table 5.4 K_d values for cesium as a function of cation exchange capacity or clay content low mica content soils

CEC (meg/100 g) / Clay Content (wt.%)	<3 / <4	3 - 10 / 4 - 20	10 - 50 / 20 - 60
K_d value (ml/g)	10	30	80

Source: EPA (1999b)

Table 5.5 also shows the estimated conservative K_d values for cesium based on CEC or clay content, but for systems containing high concentrations of mica-like minerals (i.e., >5% of the clay-size fraction).

Table 5.5 K_d values for cesium as a function of cation exchange capacity or clay content for high mica content soils

CEC (meg/100 g) / Clay Content (wt.%)	<3 / <4	3 - 10 / 4 - 20	10 - 50 / 20 - 60
K_d value (ml/g)	30	70	210

Source: EPA (1999b)

Table 5.6 shows the EPA (1999b) look-up-table for plutonium, which is a function of clay content (in wt.%) and dissolved carbonate (in meq/l).

Table 5.6 K_d values for plutonium as a function of soluble carbonate and soil clay values

Clay Content (wt.%)	0 - 30			31 - 50			51 - 70		
Soluble Carbonate (meg/100 g)	0.1 - 2	3 - 4	5 - 6	0.1 - 2	3 - 4	5 - 6	0.1 - 2	3 - 4	5 - 6
K_d value (ml/g)	5	80	130	380	1,440	2,010	620	1,860	2,440

Source: EPA (1999b)

The look-up-table for strontium, Table 5.7, requires knowledge of the CEC (or clay content) and pH of the system in order to select the appropriate conservative strontium K_d value. This table pertains to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (<0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.

Table 5.7 K_d values for strontium as a function of CEC, clay content, and pH

CEC (meq/100 g)	<3			3 - 10			10 - 50		
Clay Content (wt.%)	<4			4 - 20			20 - 60		
pH	<5	5 - 8	8 - 10	<5	5 - 8	8 - 10	<5	5 - 8	8 - 10
K_d value (ml/g)	1	2	3	10	15	20	100	200	300

Source:) EPA (1999b)

The look-up-table for thorium, Table 5.8, is based on plume thorium concentrations and pH. This table pertains to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (<0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.

Table 5.8 K_d values for thorium as a function of pH and dissolved thorium concentrations

pH	**3 - 5**		**5 - 8**		**8 - 10**	
Dissolved Th, *M*	**$<10^{-2.6}$**	**$>10^{-2.6}$**	**$<10^{-2.6}$**	**$>10^{-2.6}$**	**$<10^{-2.6}$**	**$>10^{-2.6}$**
K_d value (ml/g)	62	300,000	1,700	300,000	20	300,000

Source: EPA (1999b)

Table 5.9 shows the EPA (1999b) look-up-table for uranium, which is a function of clay content (in wt.%) and dissolved carbonate (in meq/l).

Table 5.9 K_d values for uranium as a function of pH

pH	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
K_d value (ml/g)	<1	0.4	25	100	63	0.4	<1	<1

Source: EPA (1999b)

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APPENDIX A

Generic SSLs

Tables A.1 and A.2 provide generic SSLs for 60 radionuclides in units of pCi/g and mg/kg respectively. These set of generic SSLs are derived using default values in the standardized equations presented in Part 2 of this document. Tables A.3 and A.4 provide a second set of generic SSLs. These were derived using the electronic version of the standardized equations in Part 2 to account for radionuclide decay and ingrowth. The default values (listed in Table A.5) are conservative and are likely to be protective for the majority of site conditions across the nation.

However, the generic SSLs are not necessarily protective of all known human exposure pathways, reasonable land uses, or ecological threats. Thus, before applying generic SSLs at a site, it is extremely important to compare the conceptual site model (see the *User's Guide)* with the assumptions behind the SSLs to ensure that the site conditions and exposure pathways match those used to develop generic SSLs (see Parts 1 and 2 and Table A.5). If this comparison indicates that the site is more complex than the SSL scenario, or that there are significant exposure pathways not accounted for by the SSLs, then generic SSLs are not sufficient for a full evaluation of the site. A more detailed site-specific approach will be necessary to evaluate the additional pathways or site conditions.

Generic SSLs are presented separately for major pathways of concern in both surface and subsurface soils. The pathways include external radiation exposure, inhalation of fugitive dusts, ingestion of homegrown produce, direct ingestion of soil, and migration to ground water.

The last two columns present SSLs for the migration to ground water pathway calculated using two different DAFs (dilution-attenuation factors). The second to last column presents generic SSL values for the migration to ground water pathway developed using a default DAF of 20 to account for natural processes that reduce contaminant concentrations in the subsurface (see Section 2.6.4).

The last column contains the generic SSLs for the migration to ground water pathway developed assuming no dilution or attenuation between the source and the receptor well (i.e, a DAF of 1). These values can be used at sites where little or no dilution or attenuation of soil leachate concentrations is expected at a site (e.g., sites with shallow water tables, fractured media, karst topography, or source size greater than 30 acres).

Generally, if an SSL is not exceeded for a pathway of concern, the user may eliminate the pathway or areas of the site from further investigation. If more than one exposure pathway is of concern, the lowest SSL should be used.

Analysis of Effects of Source Size on Generic SSLs

A large number of commenters on the December 1994 Soil Screening Guidance for chemicals suggested that most contaminated soil sources were 0.5 acre or less. Before changing this default assumption from 30 acres to 0.5 acre, the Office of Emergency and Remedial Response (OERR) conducted an analysis of the effects of changing the area of a chemically-contaminated soil source on generic SSLs calculated for the inhalation and migration to ground water exposure pathways. This analysis includes:

- An analysis of the sensitivity of SSLs to a change in source area from 30 acres to 0.5 acre
- Mass-limit modeling results showing the depth of contamination for a 30-acre source that corresponds to a 0.5-acre SSL

All equations, assumptions, and model input parameters used in this analysis are consistent with those described in Part 2 of the Technical Background Document for chemicals unless otherwise indicated. Chemical properties used in the analysis are described in Part 5 of the Technical Background Document for chemicals.

In summary, the results of this analysis indicate that:

- The SSLs are not particularly sensitive to varying the source area from 30 acres to 0.5 acre. This reduction in source area lowers SSLs for the inhalation pathway by about a factor of 2 and lowers SSLs for the migration to ground water pathway by a factor of 2.9 under typical hydrogeologic conditions.

- Half-acre SSLs calculated for 43 volatile and semivolatile contaminants using the infinite source (i.e., steady-state) models correspond to mass-limit SSLs for a 30-acre source uniformly contaminated to a depth of about 1 to 21 meters (depending on contaminant and pathway); the average depth is 8 meters for the inhalation pathway (21 contaminants) and 11 meters for the migration to ground water pathway (43 contaminants).

Sensitivity Analysis. For the inhalation pathway, source area affects the Q/C value (a measure of dispersion), which directly affects the final SSL and is not chemical-specific. Higher Q/C values result in higher SSLs. As shown in Table 2.4 (Section 2.3.2), the effect of area on the Q/C value is not sensitive to meteorological conditions, with thc ratio of a 0.5-acre Q/C to a 30-acre Q/C ranging from 1.93 to 1.96 over the 29 conditions analyzed. Decreasing the source area from 30 acres to 0.5 acre will therefore increase inhalation SSLs by about a factor of 2.

For the migration to ground water pathway, source area affects the DAF, which also directly affects the final SSLs and is not chemical-specific. The sensitivity analysis for the dilution factor is more complicated than for Q/C because increasing source area (expressed as the length of source parallel to ground water flow) not only increases infiltration to the aquifer, which decreases the dilution factor, but also increases the mixing zone depth, which tends to increase the dilution factor. The first effect generally overrides the second (i.e., longer sources have lower dilution factors) except for very thick aquifers (see Section 2.6.5).

The sensitivity analysis described in Section 2.6.5 shows that the dilution model is most sensitive to the aquifer's Darcy velocity (i.e., hydraulic conductivity x hydraulic gradient). For a less conservative Darcy velocity (90th percentile), decreasing the source area from 30 acres to 0.5 acre increased the dilution factor by a factor of 3.1 (see Table 2.8, Section 2.6.5). For the conditions analyzed, decreasing the source area from 30 acres to 0.5 acre affected dilution factor from no increase to a factor of 4.3 increase. No increase in dilution factor for a 0.5-acre source was observed for the less conservative (higher) aquifer thickness (46 m). In this case the decrease in mixing zone depth balances the decrease in infiltration rate for the smaller source.

Mass-Limit Analysis. The infinite source (i.e., steady-state) assumption is one of the more conservative assumptions inherent in the SSL models, especially for small sources. This assumption should provide adequate protection for sources with larger areas than those used to calculate SSLs. To test this hypothesis the SSL mass-limit models (Section 2.7) were used to calculate, for 43 volatile and semivolatile chemicals, the depth at which a mass-limit SSL for a 30-acre source is equal to a 0.5 acre infinite-source SSL.

The mass limit model is a simple mass-balance model that calculate SSLs based on the conservative assumption that the entire mass of contamination in a source leaches (migration to ground water model) over the exposure period of interest. This model was developed to correct the mass-balance violation in the infinite source (i.e., steady-state) models for highly soluble contaminants.

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Table A.1. Generic (no accounting for decay) SSLs for Radionuclides (pCi/g)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure.	Migration to Ground Water	
					20 DAF	1 DAF
Ac-227+D	8.62E-01	6.84E-01	8.69E+01	7.59E-02	—[b]	—[b]
Ag-108m	8.38E-01	4.13E+01	6.80E+05	1.55E-02	3.36E-01[c]	1.68E-02[c]
Ag-110m	6.85E-01	3.35E+01	6.42E+05	8.58E-03	5.22E+00	2.61E-01
Am-241	1.05E+01	3.66E+00	6.46E+02	4.04E+00	2.52E+00	1.26E-01
Am-243+D	9.92E+00	3.42E+00	6.73E+02	1.75E-01	2.52E+00	1.26E-01
Bi-207	1.73E+00	5.33E+01	8.65E+05	1.58E-02	—[b]	—[b]
C-14	1.28E-01	2.84E+02	2.57E+06	1.43E+04	4.00E+01	2.00E+00
Cd-109	7.00E-01	6.96E+01	8.29E+05	1.28E+01	3.48E+01	1.74E+00
Ce-144+D	1.36E+01	7.78E+00	1.65E+05	4.57E-01	2.11E+01	1.06E+00
Cl-36	1.59E-02	1.04E+02	7.26E+05	6.41E+01	—[b]	—[b]
Cm-243	1.14E+01	3.87E+00	6.75E+02	2.66E-01	2.59E+01	1.29E+00
Cm-244	1.30E+01	4.39E+00	7.18E+02	2.30E+03	2.59E+01	1.29E+00
Co-57	1.18E+01	2.86E+02	8.69E+06	3.14E-01	6.00E+00	3.00E-01
Co-60	7.89E-01	1.97E+01	5.07E+05	9.00E-03	6.00E-01	3.00E-02
Cs-134	6.85E-01	1.37E+01	1.10E+06	1.57E-02	1.63E+01	8.16E-01
Cs-135	5.99E+00	1.11E+02	9.76E+06	4.73E+03	1.84E+02	9.18E+00
Cs-137+D	9.41E-01	1.83E+01	1.53E+06	4.38E-02	4.08E+01	2.04E+00
Eu-152	6.47E+01	4.90E+01	2.00E+05	2.11E-02	—[b]	—[b]
Eu-154	3.78E+01	2.78E+01	1.58E+05	1.91E-02	—[b]	—[b]
Eu-155	2.03E+02	1.47E+02	1.23E+06	9.00E-01	—[b]	—[b]
Fe-55	1.21E+03	3.80E+02	2.27E+07	—[a]	1.32E+02	6.60E+00
Gd-153	2.54E+02	1.86E+02	2.77E+06	6.89E-01	—[b]	—[b]
H-3	4.51E+00	8.58E+03	3.23E+08	—[a]	8.00E+01	4.00E+00
I-129	2.19E-01	2.93E+00	2.99E+05	1.83E+01	4.60E-03	2.30E-04
K-40	1.37E-01	1.28E+01	1.76E+06	1.40E-01	—[b]	—[b]
Mn-54	1.51E+00	1.54E+02	3.09E+06	2.87E-02	3.06E+01	1.53E+00
Na-22	2.23E+00	4.03E+01	4.67E+06	1.08E-02	—[b]	—[b]
Nb-94	1.27E+01	3.87E+01	4.82E+05	1.53E-02	—[b]	—[b]
Ni-59	7.24E+01	1.08E+03	3.90E+07	—[a]	2.05E+02	1.03E+01
Ni-63	2.96E+01	4.43E+02	1.11E+07	—[a]	3.42E+01	1.71E+00
Np-237+D	7.74E-01	4.90E+00	1.03E+03	1.40E-01	9.00E-02	4.50E-03
Pa-231	6.23E-01	2.12E+00	3.99E+02	8.03E-01	—[b]	—[b]
Pb-210+D	4.09E-02	2.98E-01	1.31E+03	2.65E+01	6.70E-03	3.35E-04
Pm-147	2.27E+02	1.63E+02	1.13E+06	3.48E+03	—[b]	—[b]
Pu-238	8.33E+00	2.92E+00	5.40E+02	1.55E+03	1.56E+00	7.80E-02
Pu-239	8.09E+00	2.88E+00	5.45E+02	5.58E+02	1.56E+00	7.80E-02
Pu-240	8.09E+00	2.87E+00	5.45E+02	1.60E+03	1.56E+00	7.80E-02
Pu-241	6.18E+02	2.41E+02	5.44E+04	2.72E+04	2.81E+00[c]	1.40E-01[c]
Pu-242	8.53E+00	3.02E+00	5.80E+02	1.79E+03	1.56E+00	7.80E-02
Pu-244+D	7.41E+00	2.53E+00	6.20E+02	7.39E-02	1.56E+00	7.80E-02
Ra-226+D	6.83E-02	1.09E+00	1.57E+03	1.31E-02	3.20E-01	1.60E-02
Ra-228+D	2.46E-02	3.47E-01	3.47E+03	2.46E-02	3.20E-01	1.60E-02
Ru-106+D	7.68E-01	6.67E+00	1.78E+05	1.16E-01	3.12E+00	1.56E-01
Sb-125+D	1.95E+01	6.01E+01	9.41E+05	6.17E-02	—[b]	—[b]
Sm-147	1.18E+01	1.05E+01	2.64E+03	—[a]	—[b]	—[b]
Sm-151	6.98E+02	4.99E+02	3.72E+06	3.10E+05	—[b]	—[b]
Sr-90+D	4.92E-02	5.51E+00	1.61E+05	5.69E+00	1.92E-01	9.60E-03

Table A.1. Generic (no accounting for decay) SSLs for Radionuclides (pCi/g)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure.	Migration to Ground Water	
					20 DAF	1 DAF
Tc-99	7.04E-02	1.04E+02	1.29E+06	1.37E+03	3.73E+00	1.86E-01
Th-228+D	3.34E+00	9.81E-01	1.27E+02	1.44E-02	6.06E+00	3.03E-01
Th-229+D	1.97E+00	6.15E-01	8.07E+01	9.54E-02	6.06E+00	3.03E-01
Th-230	1.18E+01	3.93E+00	6.37E+02	1.36E+02	6.06E+00	3.03E-01
Th-232	1.06E+01	3.44E+00	4.19E+02	3.26E+02	6.06E+00	3.03E-01
Tl-204	8.53E-01	5.15E+01	7.41E+06	4.04E+01	—[b]	—[b]
U-232	1.46E+00	1.38E+00	9.31E+02	1.87E+02	2.40E-01[d]	1.20E-02[d]
U-233	5.81E+00	4.96E+00	1.57E+03	1.14E+02	2.40E-01[d]	1.20E-02[d]
U-234	5.90E+00	5.02E+00	1.59E+03	4.43E+02	2.40E-01[d]	1.20E-02[d]
U-235+D	5.77E+00	4.87E+01	1.80E+03	2.06E-01	2.40E-01[d]	1.20E-02[d]
U-236	6.24E+00	5.33E+00	1.73E+03	8.93E+02	2.40E-01[d]	1.20E-02[d]
U-238+D	4.65E+00	3.78E+00	1.94E+03	9.79E-01	2.40E-01[d]	1.20E-02[d]
Zn-65	2.29E-01	3.24E+01	3.13E+06	3.97E-02	1.80E+00	9.00E-02

— [a] Properties for this radionuclide are such that this pathway is not a concern at any soil concentration.

— [b] SSL cannot be calculated since a default K_d has not been specified for this radionuclide.

— [c] SSL calculated based on risk based limit for this radionuclide.

— [d] SSL calculated based on proposed MCL of 20 pCi/l (activity) for uranium.

Table A.2. Decay Corrected SSLs for Radionuclides (pCi/g)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure	Migration to Ground Water	
					20 DAF	1 DAF
Ac-227+D	1.34E+00	1.06E+00	1.35E+02	1.18E-01	—[b]	—[b]
Ag-108m	9.09E-01	4.48E+01	7.37E+05	1.68E-02	4.0E-01[c]	2.0E-02[c]
Ag-110m	2.08E+01	1.02E+03	1.95E+07	2.61E-01	1.6E+02	7.9E+00
Am-241	1.08E+01	3.75E+00	6.62E+02	4.14E+00	2.6E+00	1.3E-01
Am-243+D	9.93E+00	3.43E+00	6.74E+02	1.76E-01	2.5E+00	1.3E-01
Bi-207	2.25E+00	6.92E+01	1.12E+06	2.05E-02	—[b]	—[b]
C-14	1.28E-01	2.85E+02	2.57E+06	1.43E+04	4.0E+01	2.0E+00
Cd-109	1.15E+01	1.14E+03	1.36E+07	2.09E+02	5.7E+02	2.8E+01
Ce-144+D	3.62E+02	2.08E+02	4.41E+06	1.22E+01	5.6E+02	2.8E+01
Cl-36	1.59E-02	1.04E+02	7.26E+05	6.41E+01	—[b]	—[b]
Cm-243	1.61E+01	5.45E+00	9.51E+02	3.75E-01	3.6E+01	1.8E+00
Cm-244	2.19E+01	7.38E+00	1.21E+03	3.87E+03	4.3E+01	2.2E+00
Co-57	3.31E+02	7.99E+03	2.43E+08	8.80E+00	1.7E+01	8.4E+00
Co-60	3.17E+00	7.92E+01	2.04E+06	3.62E-02	2.4E+00	1.2E-01
Cs-134	6.91E+00	1.38E+02	1.11E+07	1.59E-01	1.6E+02	8.2E+00
Cs-135	5.99E+00	1.11E+02	9.76E+06	4.73E+03	1.8E+02	9.2E+00
Cs-137+D	1.30E+00	2.54E+01	2.12E+06	6.07E-02	5.7E+01	2.8E+00
Eu-152	1.28E+02	9.69E+01	3.95E+05	4.16E-02	—[b]	—[b]
Eu-154	9.86E+01	7.26E+01	4.12E+05	4.99E-02	—[b]	—[b]
Eu-155	8.65E+02	6.26E+02	5.22E+06	3.83E+00	—[b]	—[b]
Fe-55	9.35E+03	2.93E+03	1.75E+08	—[a]	1.0E+03	5.1E+01
Gd-153	7.96E+03	5.84E+03	8.69E+07	2.16E+01	—[b]	—[b]
H-3	9.29E+00	1.77E+04	6.66E+08	—[a]	1.6E+02	8.2E+00
I-129	2.19E-01	2.93E+00	2.99E+05	1.83E+01	4.6E-03	2.3E-04
K-40	1.37E-01	1.28E+01	1.76E+06	1.40E-01	—[b]	—[b]
Mn-54	3.67E+01	3.76E+03	7.51E+07	6.98E-01	7.4E+02	3.7E+01
Na-22	1.79E+01	3.22E+02	3.73E+07	8.67E-02	—[b]	—[b]
Nb-94	1.27E+01	3.87E+01	4.82E+05	1.53E-02	—[b]	—[b]
Ni-59	7.24E+01	1.08E+03	3.90E+07	—[a]	2.1E+02	1.0E+01
Ni-63	3.29E+01	4.93E+02	1.23E+07	—[a]	3.8E+01	1.9E+00
Np-237+D	7.74E-01	4.90E+00	1.03E+03	1.40E-01	9.0E-02	4.5E-03
Pa-231	6.23E-01	2.12E+00	3.99E+02	8.03E-01	—[b]	—[b]
Pb-210+D	6.29E-02	4.59E-01	2.01E+03	4.08E+01	1.1E-02	5.5E-04
Pm-147	1.80E+03	1.29E+03	8.95E+06	2.76E+04	—[b]	—[b]
Pu-238	9.36E+00	3.28E+00	6.07E+02	1.74E+03	1.8E+00	8.8E-02
Pu-239	8.10E+00	2.88E+00	5.46E+02	5.58E+02	1.6E+00	7.8E-02
Pu-240	8.10E+00	2.87E+00	5.46E+02	1.60E+03	1.6E+00	7.8E-02
Pu-241	1.17E+03	4.56E+02	1.03E+05	5.13E+04	1.0E+01[c]	5.0E-01[c]
Pu-242	8.53E+00	3.02E+00	5.80E+02	1.79E+03	1.6E+00	7.8E-02
Pu-244+D	7.41E+00	2.53E+00	6.20E+02	7.39E-02	1.6E+00	7.8E-02
Ra-226+D	6.88E-02	1.09E+00	1.58E+03	1.32E-02	3.2E-01	1.6E-02
Ra-228+D	9.15E-02	1.29E+00	1.29E+04	9.15E-02	1.2E+00	5.9E-02
Ru-106+D	1.60E+01	1.39E+02	3.70E+06	2.40E+00	6.4E+01	3.2E+00
Sb-125+D	1.47E+02	4.52E+02	7.07E+06	4.63E-01	—[b]	—[b]
Sm-147	1.18E+01	1.05E+01	2.64E+03	—[a]	—[b]	—[b]
Sm-151	7.82E+02	5.59E+02	4.17E+06	3.47E+05	—[b]	—[b]
Sr-90+D	6.89E-02	7.71E+00	2.25E+05	7.97E+00	2.7E-01	1.3E-02
Tc-99	7.04E-02	1.04E+02	1.29E+06	1.37E+03	3.7E+00	1.9E-01
Th-228+D	3.63E+01	1.07E+01	1.38E+03	1.57E-01	6.6E+01	3.3E+00

Table A.2. Decay Corrected SSLs for Radionuclides (pCi/g)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure	Migration to Ground Water	
					20 DAF	1 DAF
Th-229+D	1.97E+00	6.16E-01	8.08E+01	9.55E-02	6.1E+00	3.0E-01
Th-230	1.18E+01	3.93E+00	6.37E+02	1.36E+02	6.1E+00	3.0E-01
Th-232	1.06E+01	3.44E+00	4.19E+02	3.26E+02	6.1E+00	3.0E-01
Tl-204	4.71E+00	2.85E+02	4.09E+07	2.23E+02	—[b]	—[b]
U-232	1.68E+00	1.59E+00	1.07E+03	2.15E+02	2.8E-01[d]	1.4E-02[d]
U-233	5.81E+00	4.96E+00	1.57E+03	1.14E+02	2.4E-01[d]	1.2E-02[d]
U-234	5.90E+00	5.02E+00	1.59E+03	4.43E+02	2.4E-01[d]	1.2E-02[d]
U-235+D	5.77E+00	4.87E+01	1.80E+03	2.06E-01	2.4E-01[d]	1.2E-02[d]
U-236	6.24E+00	5.33E+00	1.73E+03	8.93E+02	2.4E-01[d]	1.2E-02[d]
U-238+D	4.65E+00	3.78E+00	1.94E+03	9.79E-01	2.4E-01[d]	1.2E-02[d]
Zn-65	7.11E+00	1.01E+03	9.72E+07	1.24E+00	5.6E+01	2.8E+00

— [a] Properties for this radionuclide are such that this pathway is not a concern at any soil concentration.
— [b] SSL cannot be calculated since a default K_d has not been specified for this radionuclide.
— [c] SSL calculated based on risk based limit for this radionuclide.
— [d] SSL calculated based on proposed MCL of 20 pCi/l (activity) for uranium.

Table A.3. Generic (no accounting for decay) SSLs for Radionuclides (mg/kg)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure.	Migration to Ground Water	
					20 DAF	1 DAF
Ac-227+D *	1.20E-08	9.48E-09	1.20E-06	1.05E-09	—[b]	—[b]
Ag-108m *	3.22E-08	1.59E-06	2.61E-02	5.96E-10	1.28E-08[c]	6.38E-10[c]
Ag-110m *	1.45E-10	7.07E-09	1.35E-04	1.81E-12	1.10E-09	5.51E-11
Am-241	3.06E-06	1.07E-06	1.88E-04	1.18E-06	7.39E-07	3.70E-08
Am-243+D *	4.98E-05	1.72E-05	3.38E-03	8.81E-07	1.26E-05	6.30E-07
Bi-207	3.81E-08	1.17E-06	1.90E-02	3.47E-10	—[b]	—[b]
C-14	2.86E-08	6.36E-05	5.74E-01	3.18E-03	9.00E-06	4.50E-07
Cd-109	2.72E-10	2.70E-08	3.22E-04	4.96E-09	1.33E-08	6.67E-10
Ce-144+D *	4.26E-09	2.44E-09	5.18E-05	1.43E-10	6.41E-09	3.20E-10
Cl-36	4.81E-07	3.14E-03	2.20E+01	1.95E-03	—[b]	—[b]
Cm-243	2.22E-07	7.51E-08	1.31E-05	5.16E-09	5.00E-07	2.50E-08
Cm-244	1.61E-07	5.42E-08	8.88E-06	2.85E-05	3.28E-07	1.64E-08
Co-57	1.40E-09	3.38E-08	1.03E-03	3.73E-11	7.20E-10	3.60E-11
Co-60	6.99E-10	1.74E-08	4.49E-04	7.97E-12	5.34E-10	2.67E-11
Cs-134	5.29E-10	1.06E-08	8.51E-04	1.21E-11	1.26E+14	6.32E+12
Cs-135	5.20E-03	9.61E-02	8.49E+03	4.11E+00	1.59E-01	7.96E-03
Cs-137+D *	1.08E-08	2.11E-07	1.76E-02	5.04E-10	4.69E-07	2.35E-08
Eu-152	3.66E-07	2.77E-07	1.13E-03	1.19E-10	—[b]	—[b]
Eu-154	1.43E-07	1.06E-07	5.99E-04	7.26E-11	—[b]	—[b]
Eu-155	4.38E-07	3.16E-07	2.64E-03	1.94E-09	—[b]	—[b]
Fe-55	5.05E-07	1.58E-07	9.45E-03	—[a]	5.48E+12	2.74E+11
Gd-153	7.21E-08	5.29E-08	7.87E-04	1.96E-10	—[b]	—[b]
H-3	4.69E-10	8.94E-07	3.37E-02	—[a]	8.40E-09	4.20E-10
I-129	1.24E-03	1.66E-02	1.70E+03	1.04E-01	2.28E-05	1.31E-06
K-40	1.96E-02	1.84E+00	2.53E+05	2.01E-02	—[b]	—[b]
Mn-54	1.95E-10	2.00E-08	3.99E-04	3.71E-12	3.98E-09	1.99E-10
Na-22	3.58E-10	6.45E-09	7.48E-04	1.74E-12	—[b]	—[b]
Nb-94	6.78E-05	2.07E-04	2.57E+00	8.18E-08	—[b]	—[b]
Ni-59	8.97E-04	1.34E-02	4.83E+02	—[a]	2.53E-03	1.27E-04
Ni-63	5.01E-07	7.51E-06	1.88E-01	—[a]	5.81E-07	2.91E-08
Np-237+D *	1.10E-03	6.96E-03	1.46E+00	1.99E-04	1.26E-04	6.30E-06
Pa-231	1.32E-05	4.50E-05	8.47E-03	1.70E-05	—[b]	—[b]
Pb-210+D *	5.37E-10	3.91E-09	1.71E-05	3.48E-07	8.80E-11	4.40E-12
Pm-147	2.45E-07	1.75E-07	1.22E-03	3.75E-06	—[b]	—[b]
Pu-238	4.87E-07	1.71E-07	3.16E-05	9.03E-05	9.15E-08	4.58E-09
Pu-239	1.31E-04	4.64E-05	8.80E-03	9.00E-03	2.50E-05	1.25E-06
Pu-240	3.56E-05	1.26E-05	2.40E-03	7.03E-03	6.86E-06	3.43E-07
Pu-241	6.00E-06	2.34E-06	5.28E-04	2.64E-04	2.70E-08[c]	1.35E-09[c]
Pu-242	2.17E-03	7.69E-04	1.48E-01	4.55E-01	3.95E-04	1.98E-05
Pu-244+D*	4.18E-01	1.43E-01	3.50E+01	4.17E-03	8.84E-02	4.42E-03
Ra-226+D*	6.92E-08	1.10E-06	1.59E-03	1.33E-08	3.26E-07	1.63E-08

Table A.3. Generic (no accounting for decay) SSLs for Radionuclides (mg/kg)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure.	Migration to Ground Water	
					20 DAF	1 DAF
Ra-228+D*	9.04E-11	1.27E-09	1.27E-05	9.04E-11	1.15E-09	5.76E-11
Ru-106+D*	2.28E-10	1.98E-09	5.28E-05	3.43E-11	9.36E-10	4.68E-11
Sb-125+D*	1.89E-08	5.83E-08	9.12E-04	5.98E-11	—[b]	—[b]
Sm-147	5.15E+02	4.56E+02	1.15E+05	—[a]	—[b]	—[b]
Sm-151	2.66E-05	1.90E-05	1.42E-01	1.18E-02	—[b]	—[b]
Sr-90+D*	3.61E-10	4.04E-08	1.18E-03	4.18E-08	1.42E-09	7.08E-11
Tc-99	4.16E-06	6.12E-03	7.60E+01	8.09E-02	2.19E-04	1.10E-05
Th-228+D*	4.07E-09	1.20E-09	1.55E-07	1.75E-11	7.27E-09	3.64E-10
Th-229+D*	9.26E-06	2.90E-06	3.80E-04	4.49E-07	2.87E-05	1.43E-06
Th-230	5.87E-04	1.95E-04	3.16E-02	6.76E-03	2.99E-04	1.49E-05
Th-232	9.70E+01	3.15E+01	3.84E+03	2.99E+03	5.66E+01	2.83E+00
Tl-204	1.84E-09	1.11E-07	1.60E-02	8.73E-08	—[b]	—[b]
U-232	6.84E-08	6.47E-08	4.36E-05	8.73E-06	1.13E-08[d]	5.64E-10[d]
U-233	6.03E-04	5.15E-04	1.62E-01	1.18E-02	2.52E-05[d]	1.26E-06[d]
U-234	9.47E-04	8.06E-04	2.56E-01	7.11E-02	3.84E-05[d]	1.92E-06[d]
U-235+D*	2.67E+00	2.26E+01	8.33E+02	9.52E-02	1.12E-01[d]	5.58E-03[d]
U-236	9.64E-02	8.24E-02	2.67E+01	1.38E+01	3.72E-03[d]	1.86E-04[d]
U-238+D*	1.39E+01	1.13E+01	5.79E+03	2.92E+00	7.20E-01[d]	3.60E-02[d]
uranium					2.40E-01[e]	1.20E-02[e]
Zn-65	2.78E-11	3.94E-09	3.80E-04	4.83E-12	2.16E-10	1.08E-11

*Note: The "+D" weight based concentrations only account for the concentration of the first isotope in a series and does not represent the total radioactivity.

—[a] Properties for this radionuclide are such that this pathway is not a concern at any soil concentration.

—[b] SSL cannot be calculated since a default K_d has not been specified for this radionuclide.

—[c] SSL calculated based on risk based limit for this radionuclide.

—[d] SSL calculated based on proposed MCL of 20 pCi/l (activity) for uranium.

—[e] SSL calculated based on proposed MCL of 20 ug/l (mass) for uranium.

Table A.4. Decay Corrected SSLs for Radionuclides (mg/kg)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure	Migration to Groundwater	
					20 DAF	1 DAF
Ac-227+D *	1.85E-08	1.47E-08	1.87E-06	1.63E-09	—[b]	—[b]
Ag-108m *	3.49E-08	1.72E-06	2.83E-02	6.46E-10	1.4E-08[c]	7.1E-10[c]
Ag-110m *	4.39E-09	2.14E-07	4.11E-03	5.50E-11	1.1E-09	5.5E-11
Am-241	3.14E-06	1.09E-06	1.93E-04	1.21E-06	7.4E-07	3.7E-08
Am-243+D *	4.99E-05	1.72E-05	3.38E-03	8.82E-07	1.3E-05	6.3E-07
Bi-207	4.95E-08	1.52E-06	2.47E-02	4.51E-10	—[b]	—[b]
C-14	2.87E-08	6.37E-05	5.75E-01	3.19E-03	9.0E-06	4.5E-07
Cd-109	4.44E-09	4.42E-07	5.26E-03	8.11E-08	1.3E-08	6.7E-10
Ce-144+D *	1.14E-07	6.52E-08	1.38E-03	3.83E-09	6.4E-09	3.2E-10
Cl-36	4.81E-07	3.14E-03	2.20E+01	1.95E-03	—[b]	—[b]
Cm-243	3.13E-07	1.06E-07	1.84E-05	7.28E-09	5.0E-07	2.5E-08
Cm-244	2.71E-07	9.12E-08	1.49E-05	4.79E-05	3.3E-07	1.6E-08
Co-57	3.92E-08	9.47E-07	2.88E-02	1.04E-09	7.2E-10	3.6E-11
Co-60	2.81E-09	7.01E-08	1.81E-03	3.21E-11	5.3E-10	2.7E-11
Cs-134	5.34E-09	1.07E-07	8.59E-03	1.23E-10	1.3E-08	6.3E-10
Cs-135	5.20E-03	9.61E-02	8.49E+03	4.11E+00	1.6E-01	8.0E-03
Cs-137+D *	1.50E-08	2.92E-07	2.43E-02	6.98E-10	4.7E-07	2.3E-08
Eu-152	7.25E-07	5.48E-07	2.23E-03	2.36E-10	—[b]	—[b]
Eu-154	3.74E-07	2.76E-07	1.56E-03	1.89E-10	—[b]	—[b]
Eu-155	1.86E-06	1.35E-06	1.12E-02	8.25E-09	—[b]	—[b]
Fe-55	3.89E-06	1.22E-06	7.28E-02	—[a]	5.5E-08	2.7E-09
Gd-153	2.26E-06	1.66E-06	2.47E-02	6.14E-09	—[b]	—[b]
H-3	9.68E-10	1.84E-06	6.94E-02	—[a]	8.4E-09	4.2E-10
I-129	1.24E-03	1.66E-02	1.70E+03	1.04E-01	2.6E-05	1.3E-06
K-40	1.96E-02	1.84E+00	2.53E+05	2.01E-02	—[b]	—[b]
Mn-54	4.74E-09	4.85E-07	9.71E-03	9.02E-11	4.0E-09	2.0E-10
Na-22	2.86E-09	5.16E-08	5.98E-03	1.39E-11	—[b]	—[b]
Nb-94	6.78E-05	2.07E-04	2.58E+00	8.18E-08	—[b]	—[b]
Ni-59	8.97E-04	1.34E-02	4.83E+02	—[a]	2.5E-03	1.3E-04
Ni-63	5.58E-07	8.35E-06	2.09E-01	—[a]	5.8E-07	2.9E-08
Np-237+D *	1.10E-03	6.96E-03	1.46E+00	1.99E-04	1.3E-04	6.3E-06
Pa-231	1.32E-05	4.50E-05	8.47E-03	1.70E-05	—[b]	—[b]
Pb-210+D *	8.25E-10	6.02E-09	2.63E-05	5.34E-07	9.4E-11	4.7E-12
Pm-147	1.94E-06	1.39E-06	9.66E-03	2.98E-05	—[b]	—[b]
Pu-238	5.47E-07	1.92E-07	3.55E-05	1.01E-04	9.2E-08	4.6E-09
Pu-239	1.31E-04	4.64E-05	8.80E-03	9.00E-03	2.5E-05	1.2E-06
Pu-240	3.56E-05	1.26E-05	2.40E-03	7.04E-03	6.9E-06	3.4E-07
Pu-241	1.13E-05	4.43E-06	9.98E-04	4.99E-04	5.2E-08[c]	2.6E-09[c]
Pu-242	2.17E-03	7.69E-04	1.48E-01	4.55E-01	4.0E-04	2.0E-05
Pu-244+D*	4.18E-01	1.43E-01	3.50E+01	4.17E-03	8.8E-02	4.4E-03
Ra-226+D*	6.97E-08	1.11E-06	1.60E-03	1.34E-08	3.3E-07	1.6E-08

Table A.4. Decay Corrected SSLs for Radionuclides (mg/kg)

Radionuclide	Ingestion of Homegrown Produce	Direct Ingestion of Soil	Inhalation of Fugitive Dusts	External Radiation Exposure	Migration to Groundwater	
					20 DAF	1 DAF
Ra-228+D*	3.36E-10	4.73E-09	4.74E-05	3.36E-10	1.2E-09	5.8E-11
Ru-106+D*	4.74E-09	4.12E-08	1.10E-03	7.13E-10	9.4E-10	4.7E-11
Sb-125+D*	1.42E-07	4.38E-07	6.85E-03	4.49E-10	— [b]	— [b]
Sm-147	5.15E+02	4.56E+02	1.15E+05	— [a]	— [b]	— [b]
Sm-151	2.97E-05	2.13E-05	1.59E-01	1.32E-02	— [b]	— [b]
Sr-90+D*	5.05E-10	5.66E-08	1.65E-03	5.84E-08	1.4E-09	7.1E-11
Tc-99	4.16E-06	6.12E-03	7.60E+01	8.10E-02	2.2E-04	1.1E-05
Th-228+D*	4.43E-08	1.30E-08	1.69E-06	1.91E-10	7.3E-09	3.6E-10
Th-229+D*	9.27E-06	2.90E-06	3.80E-04	4.50E-07	2.9E-05	1.4E-06
Th-230	5.87E-04	1.95E-04	3.16E-02	6.76E-03	3.0E-04	1.5E-05
Th-232	9.70E+01	3.15E+01	3.84E+03	2.99E+03	5.7E+01	2.8E+00
Tl-204	1.02E-08	6.15E-07	8.84E-02	4.82E-07	— [b]	— [b]
U-232	7.88E-08	7.45E-08	5.01E-05	1.00E-05	1.1E-08[d]	5.6E-06[d]
U-233	6.03E-04	5.15E-04	1.62E-01	1.18E-02	2.5E-05[d]	1.2E-06[d]
U-234	9.47E-04	8.06E-04	2.56E-01	7.11E-02	3.9E-05[d]	1.9E-03[d]
U-235+D*	2.67E+00	2.26E+01	8.33E+02	9.52E-02	1.1E-08[d]	5.6E-03[d]
U-236	9.64E-02	8.24E-02	2.67E+01	1.38E+01	3.7E-03[d]	1.9E-04[d]
U-238+D*	1.39E+01	1.13E+01	5.79E+03	2.92E+00	7.1E-01[d]	3.6E-02[d]
Zn-65	8.65E-10	1.23E-07	1.18E-02	1.50E-10	2.2E-10	1.1E-11

*Note: The "+D" weight based concentrations only account for the concentration of the first isotope in a series and does not represent the total radioactivity.

— [a] Properties for this radionuclide are such that this pathway is not a concern at any soil concentration.
— [b] SSL cannot be calculated since a default K_d has not been specified for this radionuclide.
— [c] SSL calculated based on risk based limit for this radionuclide.
— [d] SSL calculated based on proposed MCL of 20 pCi/l (activity) for uranium.
— [e] SSL calculated based on proposed MCL of 20 ug/l (mass) for uranium

Table A.5. Generic SSLs: Default Parameters and Assumptions

	SSL Pathway				
Parameter	**External Radiation Exposure**	**Inhalation of Fugitive Dust**	**Ingestion of Homegrown Produce**	**Migration to Ground Water**	**Default**
Source Characteristics					
Continuous vegetative cover		●			50 percent
Roughness height		❍			0.5 cm for open terrain; used to derive $U_{t,7}$
Source area (A)	❍	●	●	❍	0.5 acres (2,024 m^2); used to derive ACF for EXT, CPF for IHP, and L for MTG,
Source length (L)				●	45 m (assumes square source)
Source depth	❍		❍	❍	Extends to water table (i.e., no attenuation in unsaturated zone) for MTG
Soil Characteristics					
Soil texture		❍		❍	Loam; defines soil characteristics/parameters
Dry soil bulk density (ρ_b)	❍	●		●	1.5 kg/L, used to calculate ACF for EXT
Soil porosity (n)		●		❍	0.43
Vol. soil water content (θ_w)		●		●	0.15 (INH); 0.30 (MTG)
Soil pH				❍	6.8; used to determine pH-specific K_d
Mode soil aggregate size		❍			0.5 mm; used to derive $U_{t,7}$
Threshold windspeed @ 7m ($U_{t,7}$)		●			11.32 m/s
Meteorological Data					
Mean annual windspeed (U_m)		●			4.69 m/s (Minneapolis, MN)
Air dispersion factor (Q/C)		●			90th percentile conterminous U.S.
Fugitive particulate Q/C		●			90.801; Minneapolis, MN; 0.5-acre source
Hydrogeologic Characteristics (DAF)					
Hydrogeologic setting				❍	Generic (national); surficial aquifer
Dilution/attenuation factor (DAF)				●	20

● Indicates parameters used in the SSL equations.
❍ Indicates parameters/assumptions needed to estimate SSL equation parameters.
EXT = External radiation exposure
INH = Inhalation pathway
IHP = Ingestion of homegrown produce
MTG = Migration to ground water pathway

APPENDIX B

Sample Measurement Units, Activity and Mass

Introduction

Typically units of decay rate instead of mass are used to quantify the concentration of radioactive material in soil because the carcinogenic risks of exposure to soils contaminated with radioactive materials are related more to the decay rate of the material than to its mass. For example, one gram of ^{226}Ra has a decay rate (activity) of $3.7x10^{10}$ transformations (also referred to as disintegrations) per second, while one gram of ^{137}Cs has a decay rate of $3.2x10^{12}$ transformations per second. Since it is the energy emitted by the radioactive material during radioactive decay and the frequency of the decay that is usually of public health concern, and generally not the chemical properties of the radioactive material, it is more meaningful for health assessment purposes to quantify radioactive material according to decay rate. In addition, radioactive materials are detected and quantified by the type of radiation emitted and number of disintegrations (per unit time), not by their unique chemistry, as is the case for non-radioactive material. For these reasons, the concentration of radioactive material in soil is typically expressed in units of decay rate, pCi/g.

When and How to Calculate Mass

The equations presented in this TBD and the User Guide calculate SSLs for radionuclides in the traditional units of pCi/g. Mass units, however, provide insight and information into treatment selection, treatment compatibility, and treatment effiency, particularly for remedial actions involving mixed waste. For example, remediation goals expressed in mass are important for designing and evaluating treatment technologies such as soil separation, pump and treat, as well as subsurface barriers. Typically units for expressing mass in environmental media for soil and water are mg/kg for soil and mg/l for water. These mass units also can be expressed as parts per million (ppm) for soil and water, which is equivalent to mg/kg and mg/l. SSLs in pCi/g may be converted to mg/kg, and MCLs in pCi/l may be converted to their mass equivalent in mg/l, by the following equations:

$$SSL(mg/kg) = 2.8x10^{-12} * A * T_{1/2} * SSL(pCi/g)$$

$$MCL(mg/l) = 2.8x10^{-15} * A * T_{1/2} * MCL(pCi/l)$$

where $2.8x10^{-12}$ for soil or $2.8x10^{-15}$ for water is a conversion factor, A is the radionuclide atomic weight in g/mole, and $T_{1/2}$ is the radionuclide half-life in years. To put the relationship between SSLs expressed in these units into perspective, examine the SSL in mg/kg corresponding to a SSL of 1 pCi/g for a long-lived radionuclide such as ^{238}U and a relatively short-lived one such as ^{60}Co. ^{238}U has a half-life of $4.51x10^{+9}$ y, so a 1 pCi/g SSL would be equivalent to 3 mg/kg. On the other hand, a 1 pCi/g SSL for ^{60}Co, which has a half-life of 5.26 y, is equivalent to about $1x10^{-9}$ mg/kg. As noted in Attachment C, Table C.1, most radionuclides, for which generic SSLs have been calculated, have half-lives ranging from a few years to 10,000 years. Appendix A in the TBD shows that most SSLs for radionuclides are in fact less than 1 pCi/g so the equivalent SSL mg/kg values are even smaller. Therefore, at SSL levels, the masses of most radionuclides are extremely small values. Attachment D in the User Guide also provides the mass equivalent values for the MCLs.

Background Information on Using Mass

One important issue associated with using mass to characterize the quantities of radioactive material in the environment is that many elements, such as uranium, have several isotopes of the same element. For example, if one were to perform atomic absorption analysis of a water sample, and it revealed the presence of 1 mg/kg of uranium, there would be no way of knowing how much of the uranium in the sample is ^{238}U, ^{234}U, or ^{235}U, all of which are present in the environment naturally and due to anthropogenic activities. The potential public health and environmental impact of a given concentration of uranium in the environment will depend on the specific isotopes of uranium that are present, which could vary considerably depending on whether we are dealing with naturally-occurring uranium or uranium that may have been enriched in ^{235}U as part of the uranium fuel cycle or part of weapons production. It is also important to note that the same mass of each uranium isotope has significantly different levels of radioactivity. A mass of 1 mg/kg of ^{238}U has an activity of 0.33 pCi/g, while the same mass of ^{235}U has 2.1 pCi/g and 6,200 pCi/g of ^{234}U.

Also, many radioactive elements are present in the environment along with their stable counterpart. One example is potassium, which is naturally- occurring in the environment, ranging from 0.1 to 1% in limestones to 3.5% in granite. In addition, a typical 70 kg adult contains 130 g of potassium. A very small fraction (0.01%) of this potassium is the naturally-occurring radioactive isotope ^{40}K. If one were to measure the amount of ^{40}K in soil and assume that ^{40}K made up all of the elemental potassium, the mass of the elemental potassium would be underestimated by 10,000 fold.

Since the potential adverse effects of radioactive material are due to its disintegration rate, measurement of the mass of a given element present may not accurately represent the amount of radioactivity present and, therefore, its potential radio-toxicity.

Use of Mass in Remediation and Technology Selection

The measurement of the radioactivity present often will be a misrepresentation of the total mass of the given element and should not be used alone to calculate the treatment required for remediation technologies, since technologies are essentially chemical/physical. Doing so may underestimate the total mass of the given element and lead to errors in the amount of treatment or reactants required for remediation since technologies are chemically/physically based. For example, to design and implement a subsurface Permeable Reaction Wall for the Uranium isotopes described above, it would be necessary to know the total mass of the Uranium isotopes as well as the other aqueous reactive elements to calculate the equivalent amounts of sorption or precipitation reactants that would be required to remove or reduce the aqueous Uranium species from contaminated groundwater (EPA 2000a & EPA 1999a). The same considerations would be necessary for other ground water or water treatment technologies for dissolved concentrations of elements and their isotopic forms. For example in a pump and treat ground water extraction system that utilizes ion exchange (chemical separation) or reverse osmosis (physical separation), chemical mass measurements would be used to determine the amount and type of reactants materials, exchange capacity and effectiveness (EPA 1996). Much the same can be said for mobility limiting or mobility reduction technologies such as chemical solidification/stabilization treatability studies or treatments (EPA, 2000b). Also, mass measurements are important in the determination of Partition coefficients, K_d values that are essential in fate and transport, risk assessment modeling, and remediation calculations. K_d values are expressed in mass units for the inorganic elements and isotopes (EPA 1999b). Partition coefficients, K_d values, are the same value for all forms of the element and isotopes.

In summary, given that risk or exposure is the basis for remedial actions, mass measurements are often required for determining, designing and selecting a remediation technology. This contrasts with the need for radiation specific isotopic measurements required in risk and exposure analysis. Users should note the different applications and perspectives with their corresponding measurements units of mass and activity.

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